BOTANICAL PROCESSES IN URBAN DERELICT SPACES

by

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ABSTRACT

This thesis set out to investigate the processes that determine the richness and composition of plant communities on derelict land in the West Midlands. Experimental work included vegetation surveys, soil seed bank investigation, field mapping and seed rain trapping methods. Interpretation of the data involved a range of approaches including vegetation classification and ordination, comparative analysis of plant functional attributes and the development of regression models incorporating landscape and habitat variables. Derelict habitats were identified as holding a diverse array of communities at the early stages of succession which are poorly represented by current vegetation classifications. functional diversity is however much lower in pioneer communities. The majority of these species employing the expected strategies of early succession notably high reproductive capacity and seeds which are small, highly dispersive and form persistent seed banks. Dense seed banks were typically formed on sites and were dominated by a small set of consistently occurring species. Changes in seed bank density and composition were consistent with time represented both by the chronosequence of sites and increasing soil depth. Little evidence was found to suggest that diversity or species composition is linked to site connectivity related to either patch density or the presence of linear features. These findings have considerable implications for application of principles of metapopulation and island biogeography principles to urban conservation. In particular the trend for planners to designate urban wildlife corridors is questioned as being probably of no benefit to native diversity and indeed these features are identified as being instead potentially significant pathways for invasive alien species. The most important factors influencing the composition of sites were seen to be those linked closely with past and present human activity. Particularly significant are the nature of dereliction substrates and haphazard disturbances such as fire and tipping which influence vegetation succession temporally and spatially.

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CHAPTER ONE

BOTANICAL PROCESSES IN URBAN SPACES: A FRAMEWORK FOR PLANT COMMUNITY ECOLOGY RESEARCH IN CITIES.

1.1 INTRODUCTION

It is estimated that half the global human population now live within cities. Within the United Kingdom 90% of the population reside within cities and conurbations and urban zones contribute about 10% of the total land area (DETR 2001). For much of the last century, biodiversity conservation policy predominantly ignored these areas, focussing instead on designating protection status to a wide range of pristine natural and semi-natural habitats distributed within the rural landscape. Ecological research in cities concentrated primarily on the negative impacts associated with urbanisation and the consequences for biodiversity (see e.g. Gilbert 1968, 1971; Bishop *et al.* 1975; Seaward 1976, 1982; Stone *et al.* 1982). A recent study on plant distributions in the UK has also identified a positive relationship between local extinction and human population density (Thomson & Jones 1999).

However, in the last two decades there has been a shift towards a wider consideration of the urban ecosystem as a potential resource. Firstly, it is now acknowledged that enhancement of biodiversity within urban areas can have a positive impact on the quality of life and education of city inhabitants (Burgess, Harrison & Limb 1988). This gradual growth of understanding of the benefits of urban open spaces has led to a change in planning policies in order to integrate and further encourage outdoor activities and wildlife encounters within cities (Ecological Parks Trust 1982). It is hoped that these positive interactions will have the added benefit of raising public awareness regarding issues that affect their surrounding environments, and in turn facilitate the preservation of biodiversity in more natural habitats (Savard *et al.* 2000).

Secondly, and perhaps more pertinently for this project, urban habitats are increasingly now considered to be valuable for protection in their own right, containing unique and distinctive communities (Gilbert 1989; Shepherd 1994). Consequently the shortfall in their representation within the body of ecological research can no longer be accepted. Furthermore Agenda 21 of the United Nations Conference on Environment and Development (UNCED 1992) requires the member signatories to ensure the conservation of biodiversity at a local level. The European Sustainable Cities project's Final Report (1996) also contains a general aim in relation to soil, flora and fauna to increase the proportion of both natural and human-made eco-systems within cities. Responsibility lies firmly with local government authorities in cities to implement measures to achieve these requirements. The development of effective planning policy needs to integrate issues to preserve and enhance biodiversity within the framework of the existing urban environment. To meet these criteria, a wider interdisciplinary approach is not only desirable, but essential.

The Natural Environment Research Council (NERC) set up the Urban Regeneration and the Environment (URGENT) thematic programme to bring together ecological and environmental research within cities. This programme sets out to pool research from the geological, terrestrial, freshwater and atmospheric fields of science to provide a co-ordinated policy for the future management of cities. The URGENT Biodiversity in Urban Habitat Fragments project, of which this research thesis forms the botanical component, sets out to investigate the movement and persistence of species in patches of habitat within the West Midlands. The aim is to contribute towards decision support systems highlighting the dynamics of changing biodiversity within the urban mosaic, enabling planners to make better informed decisions to implement the UK's global and European commitments to the protection of biodiversity.

1.2 Interactive vs. Non interactive communities

To protect and enhance biodiversity in cities most effectively, it is essential to develop an understanding of the underlying ecological processes that are most significant within this unique context.

The traditional theory of plant community ecology (e.g. Hutchinson 1959; Tilman 1982) states that the composition and diversity of a locality will be determined by the interspecific interactions, such as competition, herbivory and predation, that occur. This theoretical

community has been termed the '**interactive**' community and can be seen to arise due to processes within a local spatial scale. The importance of these interactions on community structure has been demonstrated by numerous workers (see e.g. Connel 1983) and more recent studies have concentrated on examining their relative influences (Power 1992).

In contrast, other workers, (e.g. Cornell & Lawton 1992) have proposed far greater significance for relative recruitment potential in determining the composition and richness of communities (e.g. long distance dispersal attributes, the presence of dispersal barriers and the surrounding pool of species). In this model, the processes and conditions occur within a regional spatial scale and the significance of local biotic interactions is minimal. Examples of this form of community have been termed '**non-interactive**'.

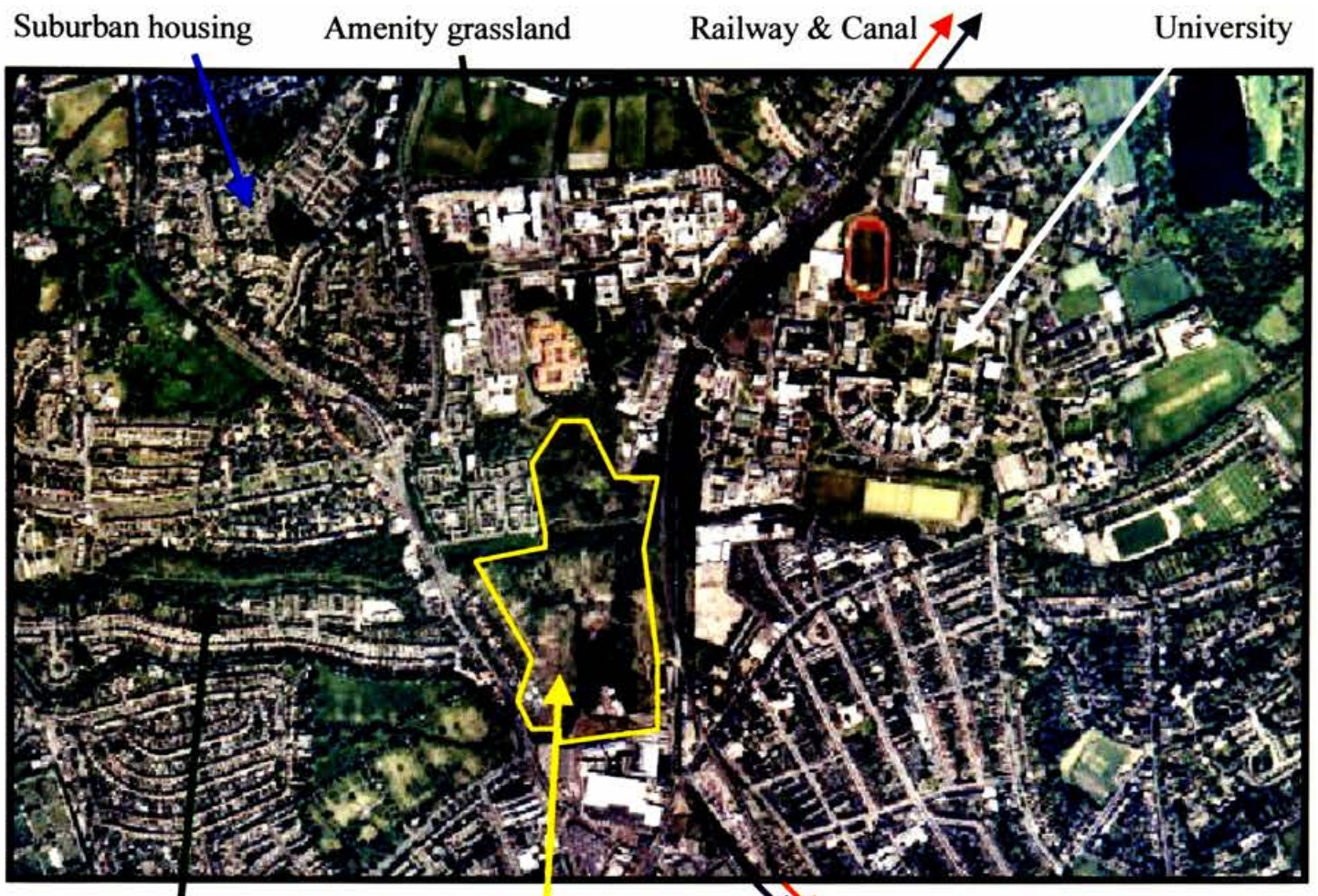
The intermediate perspective is that species richness and composition is determined by a combination of both biotic interactions and between-site dispersal factors. Consequently biological communities can be visualised as occurring at differing positions on a continuum between the interactive and non-interactive states, depending on the relative importance of each set of factors (Tilman 1982).

Within the UK, much of the natural and semi-natural vegetation of the landscape has been removed, leaving only scattered remnants distributed as isolated patches or as linear features. (Spellerberg & Gaywood 1993). The urban landscape is a complex and highly-fragmented mosaic of sparsely distributed remnant habitats and a diverse mixture of synanthropic habitats (Plate 1.1). Therefore it may be expected that recruitment limitation may be more significant in cities than it would be for communities within a continuous tract of natural or semi-natural habitat. A possible consequence of isolation due to fragmentation may be that the possible outcomes of biotic interactions are already significantly determined by the pool of species available to compete and by the relative speed of their arrival.

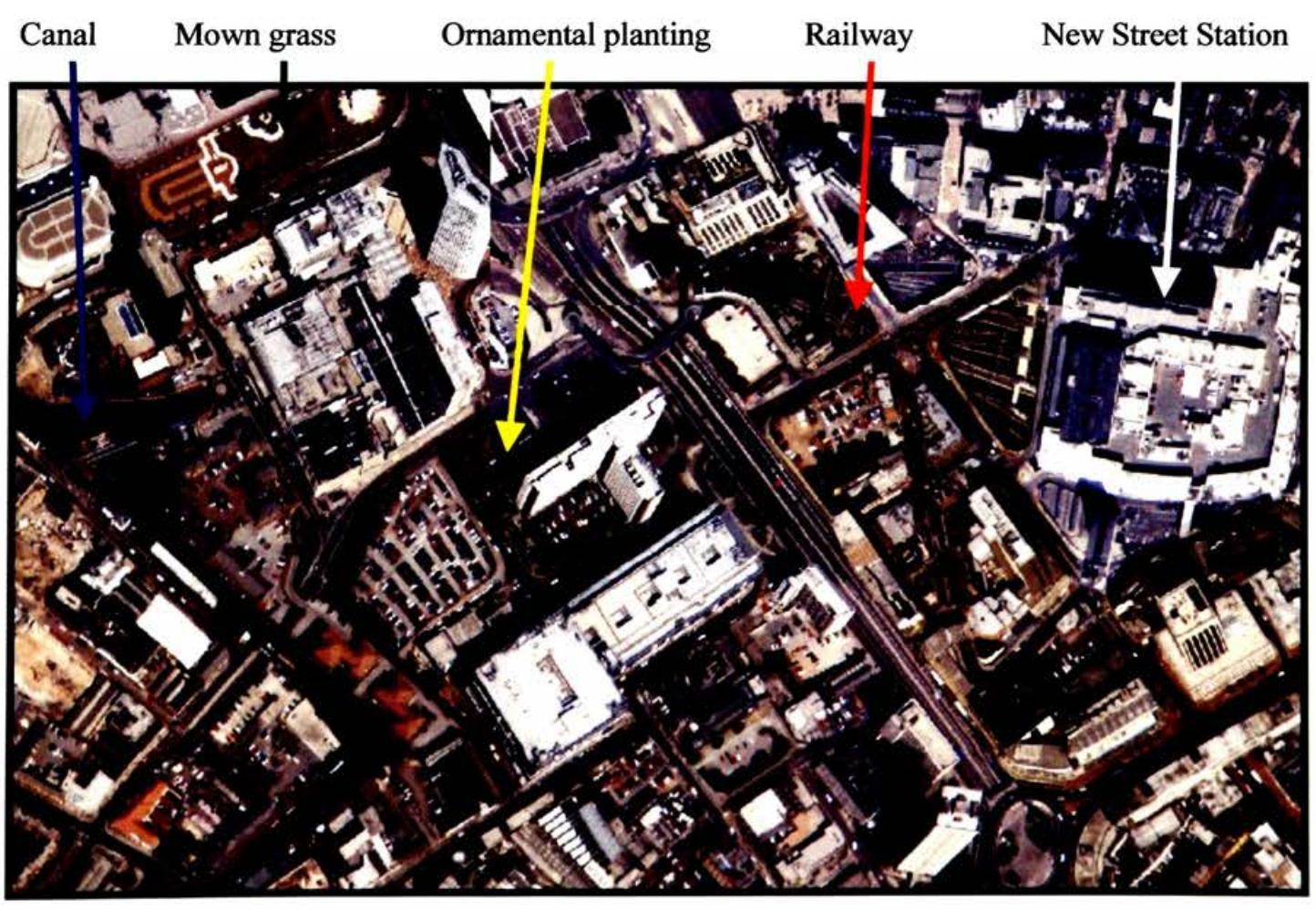
Recognition of the possible importance of spatial factors to habitats and species has been incorporated within European legislation in the last 15 years. Chapter 7 of the Final report on Sustainable Cities (European Commission 1996) considers in particular the integration of environmental and spatial planning and considers means by which environmental objectives may be identified which incorporate improved forms of public involvement in planning and the potential linkage of spatial planning and Local Agenda 21 processes.

Plate 1.1: *Views of the urban landscape in (a) suburban and (b) urban zones*

(a)



(b)



A major underlying aim of this research is to examine the relative significance of geographical isolation compared to local environmental factors to the formation of distinctive urban flora and communities. The thesis aims to consider the composition of plant assemblages based on the physiological and morphological characteristics or traits that are represented. It is expected that the relative success of different life traits and histories will vary from site to site. Because many traits show a strong correlation to the environment (e.g. Venable & Brown 1988; Thompson *et al.* 1998) and/or the dispersal potential (e.g. Salisbury 1974; Rees 1993) of a species, they can potentially provide a powerful indication of factors influencing the establishment of plant populations. If a strong relationship between species distributions and/or dispersal traits and landscape variables is found, it can be inferred that recruitment potential is of definite importance to the diversity of plants found in city habitats. Under such a scenario, the implications for planning should be to give greater consideration to the spatial context of sites. If no relationship is found, or if it is weak, then conservation should concentrate on issues specific to the actual nature of the site.

1.3 RECRUITMENT POOLS IN CITIES

Regardless of whether a site's diversity is influenced predominantly by local biotic processes or regional recruitment processes, the component species present at the site will, by definition, always reflect the regional species recruitment pool from which they are drawn. The species pool for any given region includes those species in existing growing populations and those existing in dormant stages (i.e. seed banks) that have the potential to once again proliferate. However, the limits of this recruitment pool are in fact impossible to define accurately for a number of reasons:

a) Chance climatic or biotic events (e.g. flooding, long distance animal migration) can give rise to stochastic long-distance dispersal events;

b) Some plant species, e.g. ferns and orchids, have seeds / spores which are so light they have exceptional dispersal potential carried in wind currents;

c) The direct movement of soil and plants between areas by humans is a common, but unpredictable occurrence, which has significantly altered the botanical composition of many natural ecosystems across the globe, and which has broken down natural biogeographical dispersal barriers (Elton 1952; Mooney & Drake 1986).

These limitations apply even for geographically delimited regions such as islands. For a region such as the West Midlands conurbation, which is surrounded by an artificial (human-designated) boundary, any kind of estimate would at best be an intelligent guess at a figure which is also likely to be subject to considerable temporal fluctuations. Nevertheless, for most species, the great majority of seed is deposited within a few metres of the parent plant (Verkaar 1990), and dispersal events across large distances therefore represent a tiny percentage of the total seed production. Consequently it is to be expected, that while the potential regional pool is theoretically limitless, the actual realised recruitment pool of deposited seeds is not only much smaller, but may also be significantly influenced by even small degrees of habitat isolation.

1.4 DEVELOPING AN ECOLOGICAL FRAMEWORK

The framework for this study is based on the hypothesis that it is the combination of the landscape, the environmental and the biotic spheres, which give each site its own, unique, character and which will determine the species composition found. Figure 1.1 is a schematic, highly simplified, diagram illustrating several ways in which these three spheres can interact or be interrelated with each other. There will also be strong relationships between the different factors within each sphere (e.g. vegetation & animals, climate & soil, urban rural gradients/historical land use). Examples of both types of interactions are given in the associated boxed text.

The first point made by Figure 1.1 is simply that the interactions are many and complex and that when interpreting research on any sphere, consideration must be given to the others. The second point is that each of the factors will have an effect on either or both of (i) the dispersal of diaspores to a site, and (ii) the subsequent likelihood of their establishment there.

The successful occupants of a site will be those species from the available regional recruitment pool that pass through a site-specific 'filter' based on these component factors. The criteria under which a species may or may not pass through this filter can be seen to be dependent upon their own unique set of traits and strategies which will determine both their mobility across the urban landscape and their ability to compete within the manipulated habitats encompassed within it.

Figure 1.1: *A schematic view of the interaction between biotic, environmental and landscape factors and their influence on the dispersal and establishment of plant species in the urban environment.*

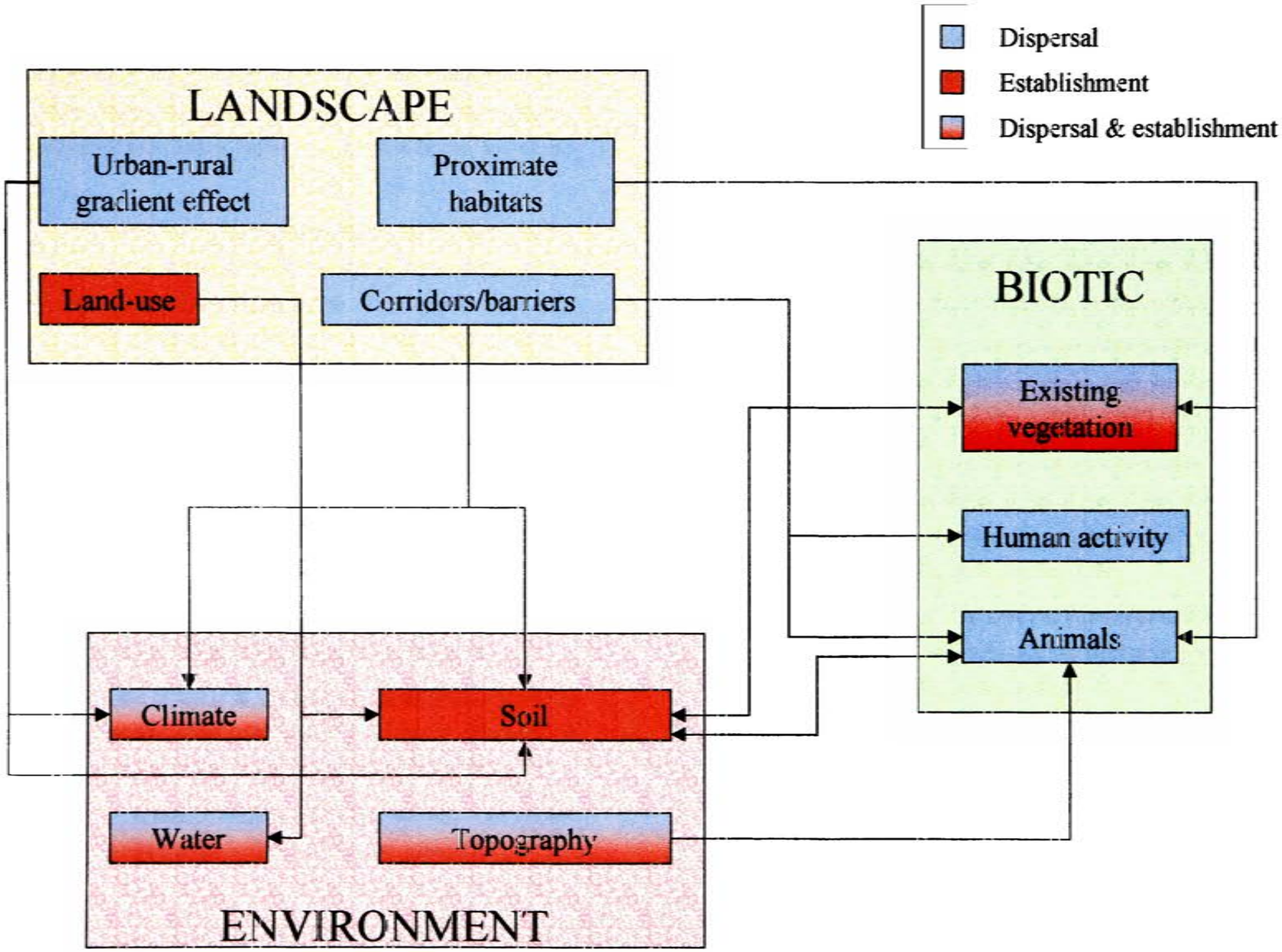


Figure 1.1: *Continued...*

Interactions within components of the urban ecosystem

The factors considered within each of the landscape, environmental and biotic components of the framework (Fig 1.1) are often closely interconnected. Each component of the urban ecosystem is considered in greater detail in later chapters but some brief examples of how factors within each component can link up are given below:

Landscape

i) associated with the gradation from rural fringe to urban centre is a change in the amount of open space and the most abundant habitats found (e.g. associated strongly with suburbia are the tracts of recreational grassland, Kelcey 1978).

ii) the most prominent linear features are often associated with former industrial land use zones as they provided routes for movement of raw materials and finished goods.

Environment

i) site topography will have a clear impact on the microclimate of a site

ii) the progress of soil formation (pedogenesis) is linked closely to climatic and topographical conditions

Biota

i) the activity of herbivorous invertebrates, primarily slugs and snails, can change the relative abundance of herbs and grasses in the vegetation (Gilbert 1989).

ii) human activities, like fire starting, can impact significantly on the flora and fauna on urban sites.

Interactions between components of the urban ecosystem

When considering the relative significance to plant communities of the a) recruitment (non interactive) and, b) (interactive) competitive, outcomes, within urban areas, the situation is complicated by the degree of overlap and interaction between the different factors which influence them.

Landscape & Environment

i) associated with the urban rural gradient is change in climate, the so called 'urban heat island effect' which can mean that the inner city can be as high as 5C warmer than the outlying edge on some nights (Chandler 1965).

ii) urban soils are often highly modified reflecting past and current land use in their structural, drainage and chemical compositions (Blockheim 1974).

Landscape & Biota

i) corridors can provide a route for dispersal for individual animals or plant seeds to travel from one population to another allowing the gradual flow of genes from one end to another (Bennett 1990)

ii) the type and location of habitat patches surrounding a site will influence the number of potential animal vectors likely to cross into a site

Environment & Biota

i) the existing flora and fauna which are found on a site are strongly influenced & determined by the underlying soil but also they continually contribute its development.

ii) topographical factors can influence the likelihood of dispersal vectors coming onto a site.

Factors influencing dispersal of species to urban habitat patches

Landscape, environmental and biotic factors may all influence the potential for a species to disperse to a given site. For example, the deposition of propagules will vary according to the nature of the receptor site. Size, shape, topography and day-to-day climatic conditions will all be significant in determining the numbers and diversity of arriving seeds.

The diversity of diaspore fragments that can be expected to reach the site will be determined by the range and proximity of habitats on surrounding land, and, at the individual species level, by the distance of populations of each taxon from the site in relation to its normal dispersal potential. However the normal dispersal range of a species, and hence the realised isolation of a site for that species, may be negatively or positively affected by a wider range of additional factors.

The matrix of land use surrounding an urban site may include large commercial and residential areas intermixed with remnant habitat patches and new habitats derived from dereliction. Habitat patches in a city are typically spatially clustered due to the interaction of its resources and the socio-economic factors that have influenced its development. High densities of semi-natural and natural communities within cities may typically be associated with areas which were unsuitable for development, e.g. along flood plains, or which were historically set aside for recreational uses, e.g. royal hunting grounds. In contrast, derelict successional habitats may be expected to be found in much higher densities in regions dominated traditionally by industries which have undergone a subsequent decline, such as armament factories and glassworks in the West Midlands.

Another distinctive feature of urban areas is the density and diversity of linear habitat features. These are often associated with current and disused transport routes (e.g. canals, railways, road verges), and with rivers and streams where these are not culverted. The nature conservation strategy for Birmingham (Land Care Associates (LCA) Ltd 1997) identifies 'key' corridors on the basis of wildlife survey reports and on their accessibility to the public. It has been suggested that such features may act as channels accelerating dispersal for some species (Bennett 1988; Dawson 1994), but may conversely act as barriers to others (Mader 1984, 1990).

Humans may also play an important role in the accidental or deliberate dispersal of plant

material across cities. The potential significance of anthropogenic activities within the modern UK landscape is high, perhaps particularly so for alien species (Hill *et al.* 1994). Large and small quantities of diaspores are moved across, into, and away from, the urban landscape by human activity on a daily basis, with the movement of garden plants, soil, garden litter and refuse. People are also often unwitting agents of dispersal carrying seeds attached to clothing or vehicles (see e.g. Clifford 1956; Hodkinson & Thompson 1997). The dispersal distance thus travelled can be a matter of metres or many miles.

A consideration of the factors which influence dispersal in the urban landscape are given in greater detail in later chapters (see PhD outline below)

Factors influencing establishment on urban habitat patches

The availability of gaps is known to be essential to the successful establishment on site (Tilman 1988, 1992) and consequently the speed of arrival will be an important determining factor influencing species establishment. In some circumstances, the existing vegetation is likely to restrict the opportunity for colonisation of newly deposited seed. Once established, stands of dominant native (e.g. *Urtica dioica*) or non-native species (*Fallopia japonica*) produce thick swards, which may reduce light radiation and strongly affect both germination and seedling development of new arrivals beneath. In addition, a number of other factors contribute to determining whether a species can successfully establish on a site.

Gilbert (1989) suggested that anthropogenic influences within cities were 'all pervading', when considering the environment within which plant communities establish. This can be considered to be both as a result of the direct effects of daily activities within the urban zone and also due to the longer-term consequences of urbanisation on the environment.

Pollution levels typically increase inwards from the rural fringe to the inner city due to the concentration of industry and the density of motor vehicle exhaust emissions associated with urban areas. Declining species diversity, associated with urbanisation, for sensitive groups has been attributed to pollution gradients (Gilbert 1968; Laundon 1973; Bishop *et al.* 1975; Grodzinska 1982). Changing growth patterns associated with pollutants have also been described for a number of urban grasses (e.g. Bell & Clough 1973; Ashenden & Williams 1980; Crittenden & Read 1978, 1979). For some species, ecological tolerance to pollution has evolved in areas suffering high deposition rates (Horsmann, Roberts & Bradshaw 1978).

In addition to direct pollution, climatic conditions also play an important role in the development of communities and the species assemblages found. Chandler (1967) demonstrated that temperature is typically significantly higher towards the centre of towns and cities, the so-called 'urban heat island effect'. Temperature is known to be a limiting factor for the spread of a number alien species and the distribution of thermophilous aliens within a city can strongly reflect the temperature gradients that occur (e.g. Sukopp & Weiler 1986). Physiological explanations for limitation may be frost intolerance (Conolly 1977) or specific growing season requirements for successful seed set or germination (for comprehensive review see e.g. Grime *et al.* 1981).

The structure, fertility and level of contamination of urban soils is related closely to previous land use. In addition, the nature of succession on urban sites is very variable depending on the primary substrate available. Where the substrate provides limited available nitrogen, for example brick rubble, legumes will provide a significant proportion of the pioneer element (Gilbert 1989). In contrast, soils on old allotments or gardens may have a structured topsoil that has been regularly manured.

Human activities also directly influence the physical structure of existing habitat patches. Fire starting, cycling, dog walking and various other activities may alter the fabric of a plant community by changing the stresses that they have to tolerate and by providing gaps for regeneration.

1.5 DERELICT LAND

Land use surveys indicate that on average 10% of urban land area is made up by derelict plots of land, much of which occurs in small patches of 1 hectare or less (DETR 2001). As habitats they are the predominant sites for ongoing succession within the urban zone. The heterogeneity of the sites due to differences in their time of abandonment, level of contamination, soil type and degree of disturbance means they have high diversity (Teagle 1978) and provide refugia for species not catered for by more managed urban green areas. They may also be a significant habitat for populations of locally rare species (Greenwood & Gemmell 1979; Shepherd 1994), particularly where they are on old industrial areas. Indeed it has been suggested that derelict sites may be essential to the continued persistence of many rare species within the British flora (Kelcey 1975).

However, derelict land is potentially a biodiversity resource that is soon to disappear. Current UK government policy foresees an increase in the amount of derelict land that is re-developed in cities. This is partly due to an urgent need for more housing. According to the latest national strategic planning policy document (DETR 2001), 60% of the 4.8 million new homes required in England and Wales by 2010 must be built upon brownfield sites, which itself will require an area of land the size of Greater London. In addition to housing, other derelict land is destined for other 'hard-end-use' redevelopment, such as business parks.

However, there are also more imaginative re-development plans that seek to transform derelict land into 'soft-end-uses', by creating public open spaces. This brings us on to the second threat to the biodiversity in derelict land: a lack of scientific understanding. Aside from the few studies on the conservation significance of derelict land, there is in general a much more considerable body of research regarding the methods by contaminated sites can be treated and by which previous habitats can be restored or new habitats created (see for reviews e.g. Dobson 1993; Land Use Consultants 1996; Hester & Harrison 1997; Cairney 1998; Bending, McRae & Moffat 1999; Gilbert & Anderson 2000; Hack 2000; Handley & Perry 2000.). However, there is still a shortfall in the amount of information about how the flora of derelict habitats is representative of the nature of the city itself and how this should be incorporated into future urban planning.

From a landscape ecology perspective, there are also advantages to using derelict sites in an investigative study. Because they are artificial habitats and not parts of remnant continuous habitats, they are invariable spatially discrete. They may therefore be treated, at least hypothetically, as islands, effectively hidden in a sea of asphalt and concrete (Crowe 1979). Therefore island biogeographical principles can, in the first instance, be considered when examining habitat patch diversity. Because of their often recent origin, derelict sites also provide an indication on the current available recruitment pool for their particular location and consequently prevents the need for considerable research to establish the nature of the surrounding landscape matrix over the period of recruitment. Consequently, derelict urban sites offer considerable scope for further research to gain further knowledge about the communities within these habitats and the processes that influence them.

1.6 RESEARCH AIMS

The research laid out in this thesis was undertaken on derelict land in the West Midlands conurbation between the summer of 1998 and autumn 1999. The broad aims of the work were to investigate patterns of species distributions on these habitat patches and to broaden understanding of what factors control the diversity of plant species found on these sites so as to inform further planning decisions in the urban zone.

Specific objectives of the thesis were to:

i) Investigate the diversity of plant communities that occur on derelict land and examine how well these can be attributed to existing vegetation associations described in the National Vegetation Classification (Rodwell 1992-99) and by Shepherd (1996).

ii) Describe the composition of the soil seed bank on urban derelict sites and compare the findings to expectations derived from studies of other habitats, in particular, with regard to the functional attributes of the component species. The wider aim was to identify the extent to which recruitment from the seed bank may play a role in maintaining biodiversity on these sites.

iii) Identify the extent to which the functional attributes exhibited by species on these sites may be related to successional age and relate the findings to the causal factors which select for different plant strategies on derelict land.

iv) Examine the relative importance of environmental and spatial factors to determining the species assemblages found on sites, their species richness and their functional composition. A particular aim of the project was to determine the significance of different measures of habitat patch isolation and also to identify whether linear features enhanced connectivity in an otherwise fragmented landscape.

A brief outline of the sections of the study follows:

Chapter 2 examines the diversity of species assemblages found on urban derelict patches and sets out to classify these associations following the methods of other workers on urban flora in the UK. The importance of the underlying substrate and succession to the communities found is investigated. An overview of the practical difficulties and potential pitfalls incurred when attempting to classify often highly heterogeneous stands of vegetation is given.

Chapter 3 provides a functional approach to understanding the changes in species composition that occur on a site as it undergoes succession. The correlation between certain plant traits and site age is examined and species are placed into functional groups according to their similarity of significant characters. The relationship between the prevailing environmental conditions of a site and the representation of species characters is investigated using these functional groups.

Chapter 4 is a comprehensive study of the seed banks found under derelict sites. Patterns of seed distribution in the soil and the similarity exhibited with the above-ground vegetation were analysed and compared to the findings expected according to hypotheses drawn from existing seed bank research. The type, frequency and intensity of disturbances typical of theses sites is explored in respect of their impact on seed bank ecology.

Chapter 5 is a critical review of the scientific evidence to substantiate the increasing designation of linear features in urban areas as wildlife corridors. This review considers the specific nature of the urban landscape and urban corridors in relation to their potential they offer as pathways for species dispersal. Consideration is also given to how the functional characteristics of species might enable predictions of the likely benefactors of corridors. The potential of these linear features to act as corridors for alien species is also discussed.

Chapter 6 investigates the importance of derelict patch connectivity when predicting which species may be found on sites. In particular, analysis attempts to identify the significance of patch density and linear features while controlling for environmental factors.

Chapter 7 considers the relative abundance of native and alien species across the conurbation in relation to landscape measures and in particular to measures of the urban-rural gradient. The results are compared and contrasted with findings from central Europe and explained by reference to the importance of point and time of arrival and by ecological constraints.

Chapter 8 examines the relative ability of differing dispersal or environmentally-related hypotheses to explain higher species richness at the edge of derelict habitat patches. A seed trapping method was designed to measure deposition from the seed rain and Ellenberg values and species traits are used as indirect measures of environment and disturbance.

Chapter 9 attempts to provide an overview of the findings of the research undertaken and to highlight the key interactions determining species distributions in cities. The implications of these findings are also discussed in relation to future urban planning and conservation policy.

CHAPTER TWO

THE FLORA OF DERELICT LAND IN AN URBAN ZONE, THE WEST MIDLANDS CONURBATION, UK

SUMMARY

The classification of urban vegetation types within national botanical surveys has been an established procedure within much of continental Europe for the last half century. By contrast, the synanthropic flora within cities in the UK has been poorly studied until recent years. The release of volume 5 of the National Vegetation Classification (ed. Rodwell 1999) and another study on the derelict flora of the Midlands, using the Zurich-Montpellier method (Shepherd 1992), provides a basis for further urban vegetation studies in Britain. This study examines the flora of 50 sites on derelict land in the West Midlands to assess how well the communities found match previous classifications. Sites were selected to ensure a broad representation of different urban zones and previous land uses. The sites also represented a chronosequence of 2-20 years since their dereliction. Variation in plant communities was examined on different substrates and across the chronosequence. Goodness of fit to previous UK classifications of urban vegetation was determined using an amended version of TABLEFIT 1.0 (Hill 1996) for NVC, which was customised to incorporate Shepherd's urban associations. Pioneer stands of vegetation proved difficult to classify due to their patchiness but intermediate tall herb communities, later successional grassland and understorey woodland communities were well matched to existing classifications. Substrate appeared to have a significant influence on the plants found at younger sites and on the speed of succession but on older sites.,vegetation had converged and was dominated by *Arrhenatheretum* grassland and *Rubus fruticosus-Holcus lanatus* understorey. The implications for urban conservation are discussed.

KEYWORDS: *National Vegetation Classification, Urban ecology, Derelict land, Succession.*

2.1 INTRODUCTION

The landscape of lowland Britain has been undergoing a largely unrecognised revolution. Large areas of previously farmed land are being set aside in the wake of overproduction and tighter quota controls. Cities have borne witness to a gradual decline in traditional industrial and manufacturing land use. Low grade and temporary housing has also been demolished on a large scale in the last 25 years. In their place, high technology companies, service sector providers and modern housing developments compete for the newly available land. Current government guidelines require 60% of new developments to occur upon brownfield sites as a measure to preserve green spaces within both urban and rural areas (*DETR 2001, 4.16-7*). However, there is considerable evidence to support the view that the loss of communities of plants and animals found upon derelict sites would be significantly detrimental to the biological diversity at both regional and national level. Rare species have been recorded as finding suitable refuge on derelict land (Greenwood & Gemmell 1978), as their natural habitats have become scarce. Indeed, 36% of the 87 species classified as rare from a comprehensive study of the flora of Nottingham were found amongst the ruderal or pioneer communities on wasteland (Shepherd 1994). High levels of environmental heterogeneity and consequently niche availability contribute to the high species richness recorded and the diversity of vegetation types present. Many sites hold communities of plants and animals that bear no reflection to those found in remnant natural habitats, or indeed the surrounding countryside (Maurer *et al.* 2000). These unique assemblages occur due to the considerable modifications made to the environment in which they grow by both former and current human land use.

Inevitably the regeneration of cities means many derelict sites with conservation importance are being redeveloped, perhaps before their value is even recognised. However local authorities are showing increased willingness to designate sites of interest as local nature reserves and the influx of lottery funding into local wildlife trusts also provides some potential for site acquisition. Consequently greater ecological understanding of the processes and factors that influence the composition of these brownfield sites is both desirable and potentially beneficial to conservation in the UK. Designation of their relative potential as pools of diversity within the urban landscape does pose problems however, as the vegetation

found growing upon them represents a range of early to mid-successional communities which are subject to considerable yearly change in species composition. In the past, the high density and rapid turnover of sites ensured diversity of successional stages represented across a city. It is this representation of different successional states in particular that ensures considerable species diversity across the city. Therefore, strategies to protect and conserve such sites must consider likely temporal changes (Hatton & West 1987). An understanding of how the mechanisms of succession operate at sites of different origin is therefore an important component for developing coherent guidelines for incorporating derelict land into urban conservation programmes.

Classification of urban vegetation

The first attempts to classify stands of vegetation in urban areas were conducted in continental Europe (Braun-Blanquet 1932; Tuxen 1950) and it is also within this region that most of the subsequent work on this subject has taken place. Synanthropic communities, those growing on habitats associated with human activity, that have been classified in former Czechoslovakia include those growing on slag and flue dust (Hejny 1971); on railways (Kovar and Leps 1986; Jehlik 1986); and within settlements ranging from small villages through to all the major cities (e.g. Kornas 1978; Pyšek & Pyšek 1990; Kubikova 1990; Pyšek 1993). In Poland, a special journal edition is dedicated to the description of urban habitats and associated communities of Warsaw (ed. Zimny 1984). Indeed cities within the former Soviet bloc are amongst the most widely studied. An overview of these East European findings can be found in Mucina (1990).

It is only recently that concerted attempts have been made to classify these unique urban communities found growing on derelict land in the UK. Early studies of communities found growing in urban situations were typically conducted on a particular substrate. Jones (1956) examined the flora of bomb sites in London after the Second World War, while both Haigh (1980) in Birmingham, and Clemens *et al.* (1984) in Sheffield, concentrated on derelict building plots. Shimwell's conspectus of urban vegetation types (published in Barret 1987) dealt briefly with specific ruderal communities. Gilbert's (1989) wider study of urban ecology also gave some consideration to repeated associations and distinctions between assemblages of different cities.

Shepherd's (1992) approach to a complete description of the major plant communities on derelict land in the Midlands was the first comprehensive attempt in Britain at classifying the synanthropic flora of cities. Utilising the Zurich-Montpellier technique he was able to relate vegetation data to existing continental classifications (Tuxen 1950) of urban flora communities. Despite occasional replacement or omission of some continental component species, explained by the isolation of Britain on the Atlantic fringe of Europe, he was able to identify 17 communities that occurred at some frequency in the region. The fifth and final volume of the National Vegetation Classification (ed. Rodwell 1999) also classifies what are defined as weed and inundation communities within the open vegetation category. Although the associations recognised within this publication are particularly typical of farmed land, several of the ruderal communities are also associated with urban situations.

Aims of the study

The study below sets out to develop a representative data set of urban derelict vegetation within the West Midlands and to examine the diversity of communities present. In particular, the aim is to investigate how adequately the existing descriptions and associations made for urban flora in the UK can account for the vegetation found on these sites. The diversity of communities is related to how substrata influences the progress and outcome of successional processes.

2.2 METHODS

Developing a single classification for testing purposes

The first step was to bring together the classifications of Shepherd (1994) and Rodwell (ed. 1999). Comparisons could then be made, and overlaps identified. The resulting combined classification system would then allow a single approach to the analysis to be adopted.

It was possible to achieve a phytosociological amalgamation due to the inclusion of the NVC communities within the European hierarchical system in the conspectus of vegetation types contained in the final volume of the National Vegetation Classification (ed. Rodwell 1999). Shepherd's (1992) study also placed his urban associations within this hierarchy. This can give some indication of how closely related are the communities of the different authors are.

However to provide exact information on how closely the communities match, it is necessary to compare their floristic components. This was achieved by incorporating the association tables of Shepherd (1992) within the TABLEFIT program (version 1.0, Hill 1996), which is used for identifying National Vegetation Classification communities, and includes the predominantly weed and inundation communities (OV) from the final Volume 5.0 of British Plant Communities. Goodness of fit between communities from both classifications could then be tested using both species composition and cover scores (Braun-Blanquet). On the basis of these findings (see Table 2.2), associations showing a high degree of similarity in both hierarchy and composition were subsequently treated as one for further analyses.

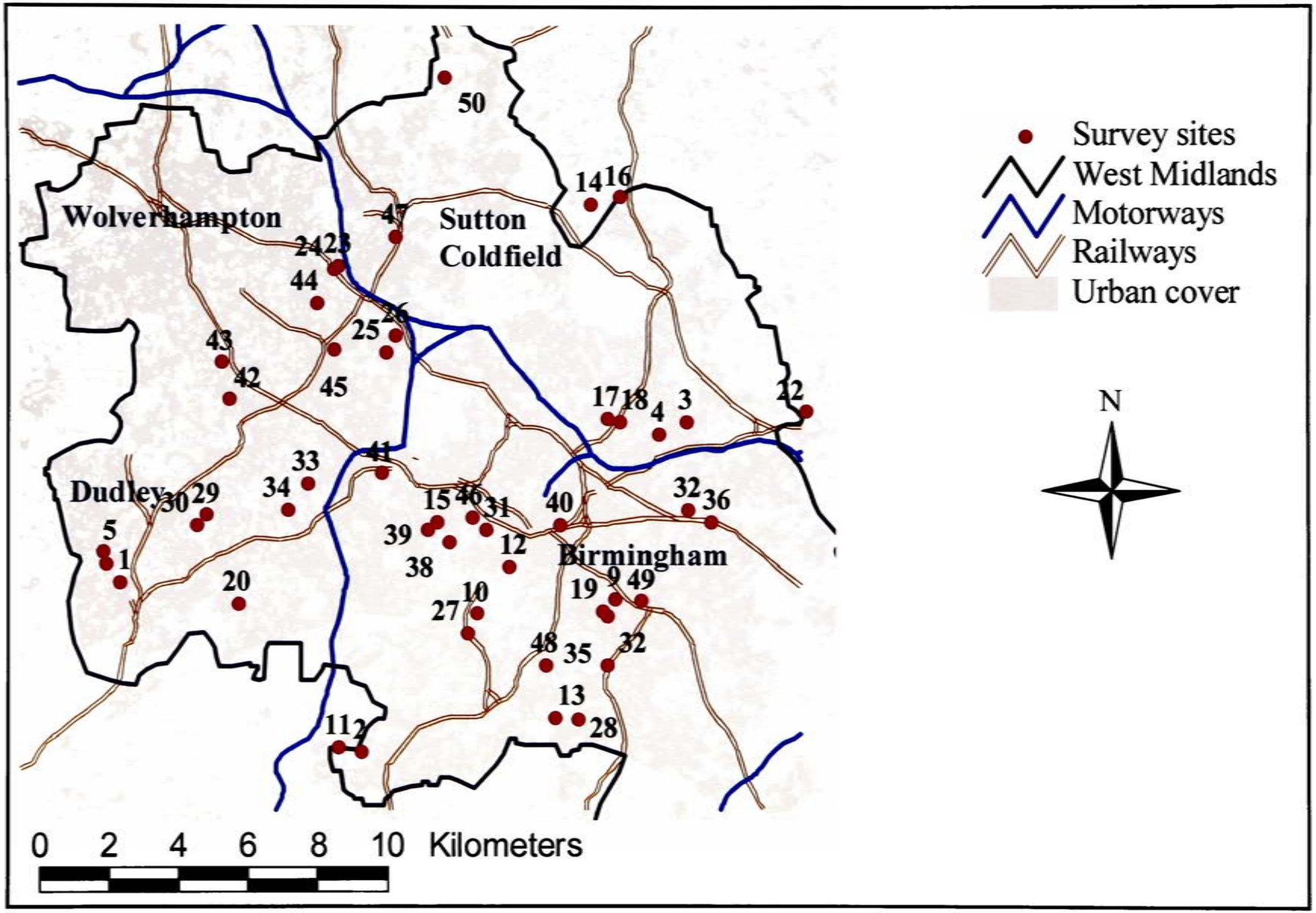
Site selection

Large continuous stands of synanthropic vegetation occur within the West Midlands in numerous situations, including road verges, cinder railway tracks and their embankments, bulldozed building sites, filled-in refuse tips or quarries and, towards the conurbation edge, arable habitats. The sampling protocol for this survey attempted to represent this diversity. Sampling was carried out at 50 derelict land sites in the West Midlands conurbation (Figure 2.1). Sites were selected either from reconnaissance in the survey region or from the current derelict land database of the local councils' planning teams. Site selection ensured a relatively even geographical spread of sites and that inner city, suburban and urban fringe habitats were all sampled. Sites were also selected to represent a chronosequence of age since dereliction. Site age was derived from various sources (see Chapter 4). The youngest site had been bulldozed 2 years prior to the study period, while the oldest had lain derelict for 20 years.

Field surveys

At each location, a number of one-metre x one-metre quadrats were surveyed according to site size. A total of 1,056 quadrats were surveyed between June and August in 1998 and 1999. The substrata of each quadrat were ascribed to descriptive categories e.g. brick rubble, tarmac. Higher plant species were identified and were assigned abundance scores according to the Braun Blanquet scale. Species not readily identified were collected in clearly marked sample bags for later examination. The finalised species lists and abundance categories for individual quadrats were then run through the amended version of TABLEFIT 1.0 (described above).

Figure 2.1: *Map of the fifty derelict survey sites in the West Midlands*



2.3 RESULTS (1): A UNIFICATION OF EXISTING CLASSIFICATIONS OF UK URBAN FLORA

Table 2.1 exhibits the most frequently observed communities or associations recorded for urban situations in the UK by Shepherd (1994) and Rodwell (1999); the alliances to which they belong in the European phytosociological hierarchy; and their associated habitats. The positioning of these alliances has been rearranged here to approximate their placement within a successional series based on observations by these workers. A full hierarchical breakdown including authorities derived from the conspectus of Rodwell (1999) and adapted to include Shepherds (1994) communities can be found in Appendix II.

Pioneer communities (Table 2.1a)

The most distinctive of the early successional urban alliances is Sisymbrium officinalis. The associations in this alliance are very frequent on brick rubble sites and on mounds of earth at the edges of sites with generally well-drained dry soils. More specialised early successional alliances include the Fumario-Euphorbion, in which associations are commonly recorded over highly enriched soil (e.g. as weed communities amongst ornamental plantings); and the Thero-Airon, which previously was mostly associated with xerophytic locations, particularly railway clinker. The most frequently occurring woody pioneer community is the *Epilibio-Salicetum* which is within the Sambuco-Salicion capreae and which has often been recorded in cities over coarse substrate such as broken concrete and brick rubble.

Tall herb communities (Table 2.1b)

Tall herb communities generally represent the phase of vegetation after the pioneer species in most urban successions (Gilbert 1989). The most widely recorded tall herb communities in British cities fall within the alliance Galio-Alliarion and Arction lappae in which *Artemisia vulgaris* and *Urtica dioica* are both regularly occurring species. The principal factor determining which of the different associations within these alliances is recorded has been shown to be the fertility of the site and the amount of disturbance to which it is exposed. Several communities within this alliance are particularly found over very fertile soil, and are consequently abundant within arable situations but are also recorded frequently on dumped topsoil and rich loamy soils in urban areas.

Table 2.1: *A successional breakdown of the phytosociological alliances of the most commonly vegetation found on synanthropic urban locations in Britain. Adapted to incorporate the urban classifications of Shepherd with relevant NVC communities. Sources: Shepherd 1994; Rodwell ed. 2000*

a) Early successional and heavily trampled vegetation

ALLIANCE	COMMUNITY	DESCRIPTION OF URBAN LOCATIONS
Sisymbrium officinalis	SH1 *Hordeetum murinii*	Road verges, base of walls, edges of car parks and paved areas on dry soils
	SH2 *Sisymbrietum loeselii*	Brick and earth mounds, edges of car parks
	SH3 *Sisymbrium officinale-Tripleurospermum* community	Mounds of earth, brick and concrete often on construction sites over fertile sandy loams
	SH5 *Senecio squalidus-Conyza canadensis* community	Bare patches, base of walls, over brick, gravel, concrete & railtrack. Well drained, dry soil with low organic matter
Fumario-Euphorbion	SH5 *Fumaria officinalis-Veronica persica* community	Predominantly in landscape areas growing over freshly laid topsoil
	OV13 *Stellaria media-Capsella bursa-pastoris* community	Disturbed ground enriched through dumping of topsoil
Thero-Airion	SH14 *Hieracium pilosella* community	Railway lines on banks and cinder and clinker
	SH15 *Vulpia myuros* community	Disused railway lines on loose dry soils with cinder and clinker
Sambuco-Salicion capreae	SH12 *Epilibio-Salicetum*	Derelict sites over concrete & brick rubble often with thin dry gravelly soil
Not determined	SH4 *Chenopodium-Atriplex prostata* community	Mounds of earth with brick rubble & given topsoil application. Moist fertile soils

Table 2.1 *continued*

b) Tall herb vegetation

ALLIANCE	COMMUNITY	DESCRIPTION OF URBAN LOCATIONS
Galio-Alliarion	OV24 *Urtica dioica-Galium aparine* community	Well-aerated nutrient rich soils on dumps of soil on wasteland or in unkept gardens
	OV25 *Urtica dioica-Cirsium arvense* community	Nutrient rich loamy soils on verges and waste ground
Arction lappae	SH6 *Lamio-Conietum maculati*	Rubbish tips and earth mounds and on disturbed ground in moist situations e.g. adjacent to canals or ditches
	SH7 *Tanaceto-Artemisietum*	Edges of derelict plots, car parks, and roadsides on dry poor to moderately fertile soil
	SH9 *Artemisia vulgaris-Urtica dioica* community	Earth and rubble mounds on derelict building plots and on road verges. Sandy loams mixed with brick, concrete and rubbish
Rumicion obtusifolii	SH10 *Rumicietum obtusifolii*	Mounds of earth on construction sites or on sides of the road over sandy loams
Dauco-Melilotion	SH11 *Melilotetum albae-officinalis*	On derelict land, clay pits & railway sidings. Sandy loams with brick rubble or concrete
Carici piluliferae-Epilobion angustifolii	OV27 *Epilobium angustifolium* community	Railway embankments, wasteland, old fire patches, over fertile soil

Table 2.1 *continued*

c) Trampled vegetation

ALLIANCE	COMMUNITY	DESCRIPTION OF URBAN LOCATIONS
Polgonion avicularis	OV18 *Polygonum aviculare-Chamomilla suaveolens* community	Paths over loamy and sandy soils on wasteground
	OV19 *Poa annua-Matricaria perforata* community	Road verges
	OV20 *Poa annua-Sagina procumbens* community	Cracks in well kept pavements and walls, often the only vegetation type where present
	SH13 *Polygono-Matricaria matricarioidis*	Road verges, gardens, landscaped areas prone to disturbance. Fertile loamy sands
Polygono-Chenopodion polyspermii	OV9 *Matricaria perforata-Stellaria media* community	Road sides
	OV10 *Poa annua-Senecio vulgaris* community	Trampled ground, on dumped earth, gardens, recreation fields
	OV12 *Poa annua-Myosotis arvensis* community	Trampled recreational areas
Lolio-Plantaginion	MG7 *Lolium perenne* leys and related grasslands	Recreational areas & damp road verge
	OV21 *Poa annua-Plantago major* community	Heavily trampled areas such as tracks and paving in recreation areas, wasteland and road sides
	OV22 *Poa annua-Plantago major community*	Disturbed, lightly trampled paths, beside walls, pavements, verges, gardens & wasteland
	OV23 *Poa annua-Taraxacum officinale* community	Sown recreation areas, playing fields, some disturbance and low frequency mowing

Table 2.1 continued

d) Open grassy swards

ALLIANCE	COMMUNITY	DESCRIPTION OF URBAN LOCATIONS
Cynosurion cristati	MG5 *Cynosurus cristatus-Centaurea nigra* grassland	Road verges, railway embankments and old quarries
	MG6 *Lolium perenne-Cynosurus cristatus* grassland	Recreation swards, lawns and road verges on moist free draining soils
Elymo-Rumicion crispi	MG11 *Festuca rubra-Agrostis stolonifera-Potentilla anserina* grassland	Moist but free draining soils alongside ditches and roadsides
	OV28 *Agrostis stolonifera-Ranunculus repens* community	Waterlogged clay waste ground and muddy tracks

e) Coarse grassy swards and understorey vegetation

ALLIANCE	COMMUNITY	DESCRIPTION OF URBAN LOCATIONS
Calthion palustris	MG9 *Holcus lanatus-Deschampsia cespitosa* grassland	Waste ground most frequently where soil conditions are anaerobic due to waterlogging
Arrhenatherion elatioris	MG1 *Arrhenatherum elatius* grassland / SH16 *Arrhenatheretum elatioris*	Widespread. Road verges, embankments, churchyards, waste ground, dumps, disused quarries on a variety of substrata
Rubion subatlanticum	W24 *Rubus fruticosus-Holcus lanatus* understorey	Abandoned and neglected land particularly derelict sites and disused gardens on a variety of substrata

In the alliance Dauco-Melilotion, the *Melilotetum albae-officinalis* association (which regularly includes all three of the common urban *Melilotus* spp., *M.officinalis*, *M.alba* and *M.altissima*) is particularly frequent in cities over brick rubble or clay infill, and alongside railways. Another alliance containing tall herb vegetation is the Rumicion obtusifolii, which has been noted to be particularly found on mounds in areas where construction is taking place. Also included within this vegetation phase is the *Chamerion (Epilobium) angustifolium* community within the alliance Carici piluliferae-Epilobion angustifolii, which is apparently the most frequent community to develop after fire.

Trampled vegetation (Table 2.1c)

A number of associations familiar in cities show tolerance to trampling, which essentially acts to inhibit the successional progression of the vegetation that would otherwise occur. Vegetation of this kind is very frequent in rural areas alongside waysides, farm tracks and areas where animals gather. In cities, the alliances to which these communities belong are found in areas subject to considerable human activity. The most important of these alliances are the Polgonion avicularis, the Polygono-Chenopodion polyspermii and the Lolio-Plantaginion. The communities within them are found on a variety of habitats and locations but are particularly common on road sides, gardens, footpaths and heavily used recreational areas.

Open grass swards (Table 2.1d)

Tall herb communities are typically succeeded by grassy swards, that are initially quite open and somewhat low growing. The communities found at this stage of the succession are primarily determined by the amount of moisture retained by the soil. Over free-draining substrates, the most frequently recorded associations are those containing a high cover of *Cynosurus cristatus* and *Lolium perenne* within the alliance Cynosurion cristati. Where the soil is moist or even waterlogged, the vegetation more often falls within the Elymo-Rumicion crispi alliance where *Agrostis stolonifera* and *Festuca rubra* are particularly abundant.

Coarse grass swards and later successional vegetation (Table 2.1e)

With increasing deposition of biomass from the open grassy swards there is a gradual transition to thicker grass swards, which can again be largely predicted by the underlying

moisture of the substrate. Associations within the alliance Calthion palustris are primarily found over damper and particularly waterlogged soils with anaerobic conditions. The coarse-bladed tussocks of *Deschampsia caespitosa* may become the dominant species in this type of vegetation. Outside of these highly waterlogged situations, vegetation within the alliance Arrhenatherion elatioris is almost ubiquitous on waste ground and road verges that have been neglected for some time, as it has a wide tolerance of soil types. The most recorded community that occurs as succession progresses from these coarse grassland types is *Rubus fruticosus-Holcus lanatus* understorey within the alliance Rubion subatlanticum.

Overlap between Shepherd's synanthropic communities and NVC communities

Table 2.2 illustrates the NVC communities that show the best fit to the associations of Shepherd's (1992) study, based on the output from TABLEFIT analyses. Four of the communities showed a large degree of overlap. The *Fumaria officinalis-Veronica persica* community was found to be closely equivalent to the NVC's lowland arable weed community OV7 *Veronica persica-polita*. The *Polygono-Matricaria matricoides (Tripleurospermumu inodorum)* community fitted closely to the *Lolium perenne-Capsella bursa-pastoris* subcommunity of OV19b *Poa annua-Tripleurospermum inodorum*. The *Lamio-Conietum maculat*i was found to be closely similar to OV25 *Urtica dioica-arvense* in species composition, although different in terms of the degree of dominance exhibited by *Urtica dioica*. The *Arrhenatheretum elatior* corresponds closely with MG1 *Arrhenetheretum elatioris*, and also is similarly placed within the phytosociological hierarchy (Table 2.1). Each of these four pairs of closely overlapping communities were treated together in further analyses.

Five communities were found to be very distinct from any NVC classification, having no matches that showed more than a 30% fit. These were the *Hordeetum murini*, *Senecio squalidus-Conyza canadensis*, *Melitotetum albae-officinalis*, *Hieracium pilosella* and *Vulpia myuros* communities.

Table 2.2: *Similarity of Shepherd's (1992) communities to National Vegetation Classification scheme. Comparison was made using TABLEFIT 1.0 (Hill 1996) and where applicable total Goodness of Fit is given (%)*

ID	SHEPHERD'S COMMUNITY TYPE	COMPARISON WITH NVC
SH1	*Hordeetum murini*	Very distinct
SH2	*Sysimbrietum loeselii*	Weak similarity (59%) to OV19b *Poa annua – Tripleurospermum inodorum*
SH3	*Sysimbrium officinalis-Tripleurospermum inodorum* community	Very weak similarity (49%) to OV19b *Poa annua – Tripleurospermum inodorum*
SH4	*Chenopodium album-Atriplex prostata* community	Similarity (52%) to OV7 *Veronica persica-polita*
SH5	*Fumaria officinalis-Veronica persica* community	Strong similarity (81%) to OV7 *Veronica persica-polita*
SH6	*Senecio squalidus-Conyza canadensis* community	Very distinct
SH7	*Lamio-Conietum maculati*	Similar (75%) to OV25 *Urtica dioica-Cirsium arvense*
SH8	*Tanaceto-Artemisietum*	Weak similarity (68%) to OV25 *Urtica dioica-Cirsium arvense*
SH9	*Artemisia vulgaris-Urtica dioica* community	Weak similarity (59%) to OV25 *Urtica dioica-Cirsium arvense*
SH10	*Rumex obtusifolius* community	Weak similarity (63%) to OV25 *Urtica dioica-Cirsium arvense*
SH11	*Melitotetum albae-officinalis*	Very distinct
SH12	*Epilobio-Salicetum caprea*	Very weak similarity (49%) to OV27b *Epilobium/Chamerion*
SH13	*Polygono-Matricaria matricoides*	Similar (77%) to OV19b *Poa annua-Tripleurospermum inodorum* and OV10b *Poa annua-Senecio vulgaris*
SH14	*Hieracium pilosella* community	Very distinct
SH15	*Vulpia myuros* community	Very distinct
SH16	*Arrhenatheretum elatior*	Similar (75%) to MG1 *Arrhenatheretum elatioris*
SH17	*Holcus lanatus* transition community	Very weak similarity (46%) to OV23d *Lolium perenne-Dactylis glomerata*

2.4 RESULTS (2): THE DISTRIBUTION OF URBAN VEGETATION TYPES ACCORDING TO SUBSTRATE AND SITE AGE IN THE WEST MIDLANDS.

Classification of West Midlands quadrats in relation to age and substrate

Table 2.3 illustrates the goodness of fit (GOF) between the 1056 survey quadrats and their closest matching community. More than a quarter of the quadrats showed very poor fits (<50%), and consequently these were excluded from further analyses.

Shepherd's urban types were well-represented (26% of quadrats, Figure 2.2) as were NVC mesotrophic (MG) grassland (33%) and woodland understorey (W) (13%). In contrast, NVC weed and inundation (OV) communities provided fits for only 19% of quadrats, and in many instances this fit was weak. 44% of quadrats classified in the 'other' NVC category were found to have fits that were weaker than 50%. This reflects the mismatching of associations to inappropriate communities at very low similarity.

Pearson's product moment correlation (p.m.c.c.) was used to investigate the relationship between site age and the goodness of fit between survey quadrats and existing vegetation classifications (Figure 2.3). This demonstrated a significant positive correlation indicating that younger quadrats tended to be less well matched.

Tables 2.4a & b illustrate the sites at which classification attempts were most, and least successful, respectively. The results provide further evidence that classification of vegetation at younger types is less successful. Bare ground sites, including both arable old fields and young derelict urban sites, often showed very poor matching. By contrast, quadrats on sites where succession has progressed to a thicker sward of dominant perennial grasses tended to show significantly better matches with described communities.

GOF was also found to be influenced by substrate (Figure 2.4). The most frequently encountered substrates in this study (concrete or tarmac, clay infill, brick rubble and urban topsoil) were all found to have vegetation that provided at least reasonable matches in between 70-80% of cases. By contrast, in quadrats where sand or ballast was the underlying material, the vegetation often showed very poor matching. Quadrats from old arable fields also showed poor similarity to any community (Table 2.4b).

***Table 2.3**: Goodness of Fit (GOF) between all survey quadrats and recognised plant communities as tested using an amended version of TABLEFIT 1.0 (Hill 1996).*

GOODNESS OF FIT (% SIMILARITY)	RATING	NUMBER OF QUADRATS	% OF QUADRATS
80-100	Very Good	70	6.6
70-79	Good	197	18.7
60-69	Fair	268	25.4
50-59	Poor	247	23.4
0-49	Very Poor	274	25.9

***Figure 2.2:** Abundance of different classification groups represented in TABLEFIT output for derelict land survey (W – woodland, MG – mesotrophic grassland, OV – open vegetation, SH – Shepherd (1996) communities, Other – Other NVC communities). Percentage reduction of quadrats fitting categorty minimum fit of 50% is applied is shown in parentheses.*

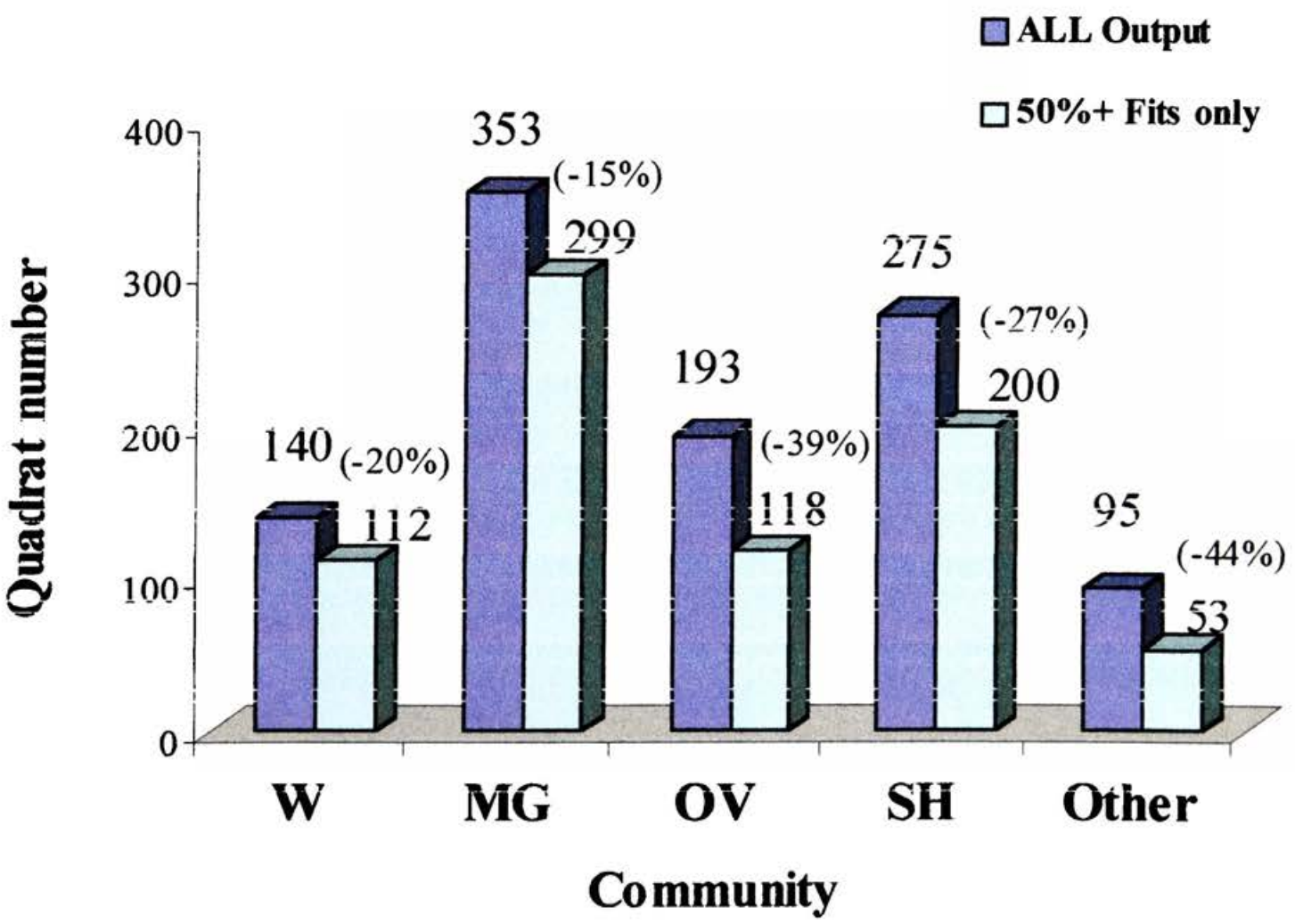


Figure 2.3: *Correlation between quadrat Goodness of Fit (G.O.F) score (fit to closest matching derelict community type) and age of site.*

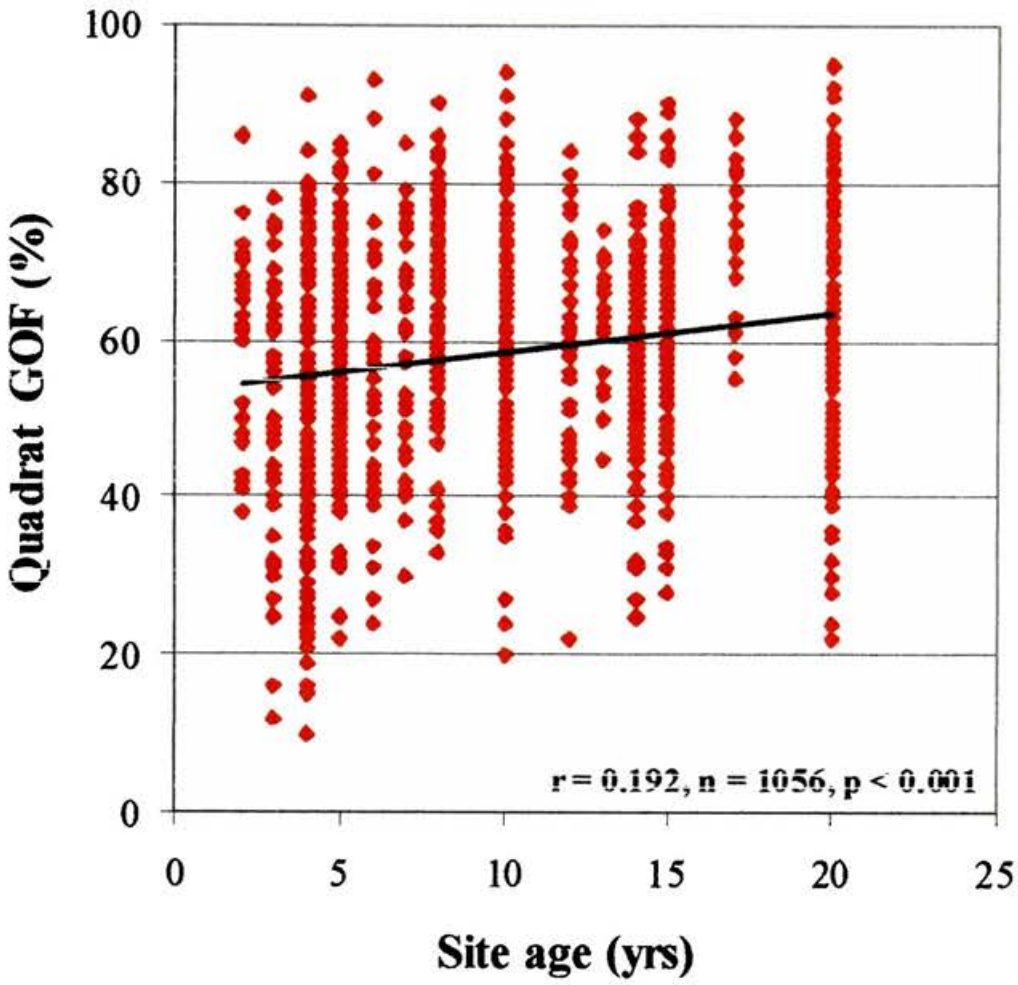


Figure 2.4: *Quadrat Goodness of Fit (G.O.F.) scores at different substrate types*

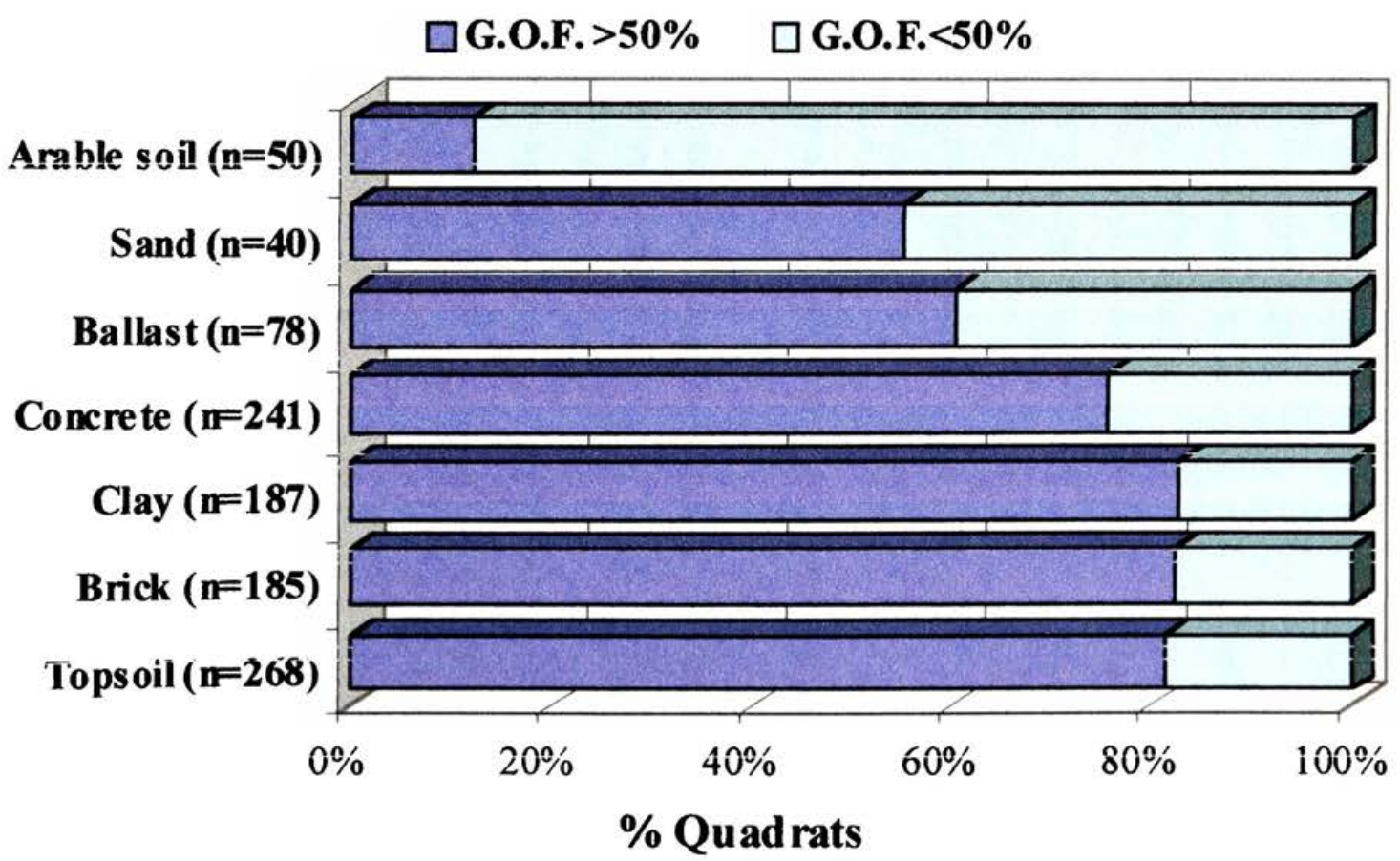


Table 2.4a: *The ten sites at which vegetation quadrats showed closest similarity to previously classified vegetation communities.*

SITE	NAME	AGE	DESCRIPTION	QUADS	% WITH G.O.F. <50%	MEAN G.O.F..
20	Cradley Heath	17	Demolished house and garden	20	0%	72.2 (± 9.6)
29	Saltwells Bus stop	20	Rough grassland adjacent. to road	15	0%	71.1 (± 13.4)
49	Tyseley Wharf	5	Raised clay embankment	20	0%	68.3 (± 10.5)
42	Foxyards Road	10	Small patch of tall herbs on corner of main road	15	0%	67.9 (± 12.6)
41	Woodlands Drive	4	Former prefab housing, now open grassland & bare ground	30	0%	67.8 (± 10)
36	Cole Roundhay	8	Former garages and garden amongst housing estate	10	0%	67.5 (± 8.8)
1	Stourbridge Housing	13	Rough grassland patch behind new housing estate	15	0%	61.6 (± 8.4)
37	Mid Cole	20	Former dumping ground, now Coarse grassland, by river	25	4%	71.8 (± 13.2)
18	Gravelly High	15	Formerly houses, now coarse grassland	20	5%	72.9 (± 9.5)
17	Gravelly Low	15	Formerly houses, now coarse grassland	20	5%	63.9 (± 9.5)

Table 2.4b: *The ten sites at which quadrats show least similarity to previously classified vegetation communities*

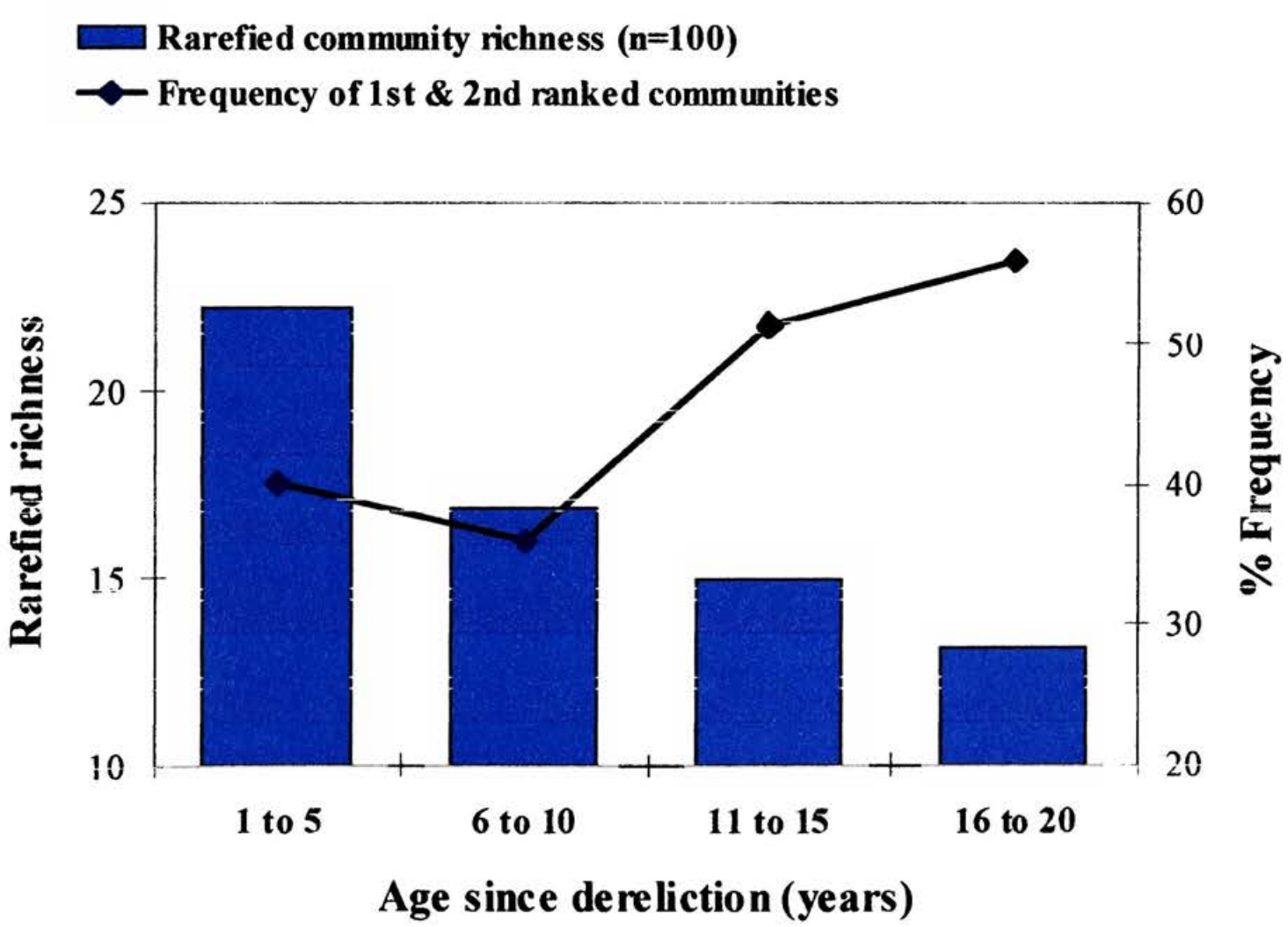
SITE	NAME	AGE	DESCRIPTION	QUADS	% WITH G.O.F. <50%	MEAN G.O.F..
8	Solihull	4	Old arable field site	20	90%	31.8 (± 10.3)
14	Roman Rd	4	Old arable field site	30	87%	36.3 (± 11.3)
5	Glass-cutters	3	Open thin vegetation on recently demolished factory site	20	80%	38.3 (± 12.6)
13	Kings Heath	5	Low growing vegn. on derelict car park w. regular bonfires	15	67%	45.3 (± 12.7)
16	Lake St	10	Road edge with a mixture of tall herbs and trampled verge	15	60%	46.4 (± 16.6)
33	Samson Quarry	20	Calcareous grassland growing over slopes of old quarry	25	60%	48.8 (± 9.2)
9	Ackers	7	Slope adjacent to ski run produced from heaped refuse	20	50%	50.9 (± 11.9)
11	Ruberry Shops	4	Mound of clay near shops	20	45%	50.8 (± 14.8)
23	Sm.Bentley Mill	6	Recently formed embankment beside overflow brook	24	42%	50.3 (± 14.2)
50	Brownhills	20	Stretch of disused railway track on embankment	15	40%	51.1 (± 16.7)

Change over time in the diversity of communities recorded

Further analysis examined the change in the diversity of communities that occurs over time following dereliction. Using only quadrats with GOF > 50%, the dataset was divided into 4 categories according to age: i) 2-5years, ii) 6-10years, iii) 11-15years, and iv) 16-20years old. To overcome the problem of unequal sample size, a rarefaction method (Sanders 1968; Hurlbert 1971) was used to compute the expected number of communities in a standardised sampling unit of 100 quadrats. Rarefied community richness and the frequency of the two most abundant communities found in each category were plotted against site age (Figure 2.5). The results suggest that over time, sites become dominated by a smaller number of communities, i.e. the vegetation found tends to converge. In the later successional states (16-20years) the most abundant two communities account for over 50% of quadrats sampled.

The changes over time in the abundance of the twelve most frequently occurring communities

Figure 2.5 *Change in community and dominance along the age gradient*



are illustrated in Figures 2.6 a-c. Figure 2.6a depicts the four most dominant communities in the study, which are all coarse grassland or understorey scrub. As was seen in Figure 2.5, the two commonest communities (MG1 *Arrhenatherum elatioris* and W24 *Rubus fruticosus-Holcus lanatus* understorey) increase in dominance with age. At the older sites, they cover more quadrats than their respective dominant communities MG11 (*Festuca rubra-Agrostis stolonifera-Potentilla anserina)* and SH17 (*Holcus lanatus* transition) cover at younger sites.

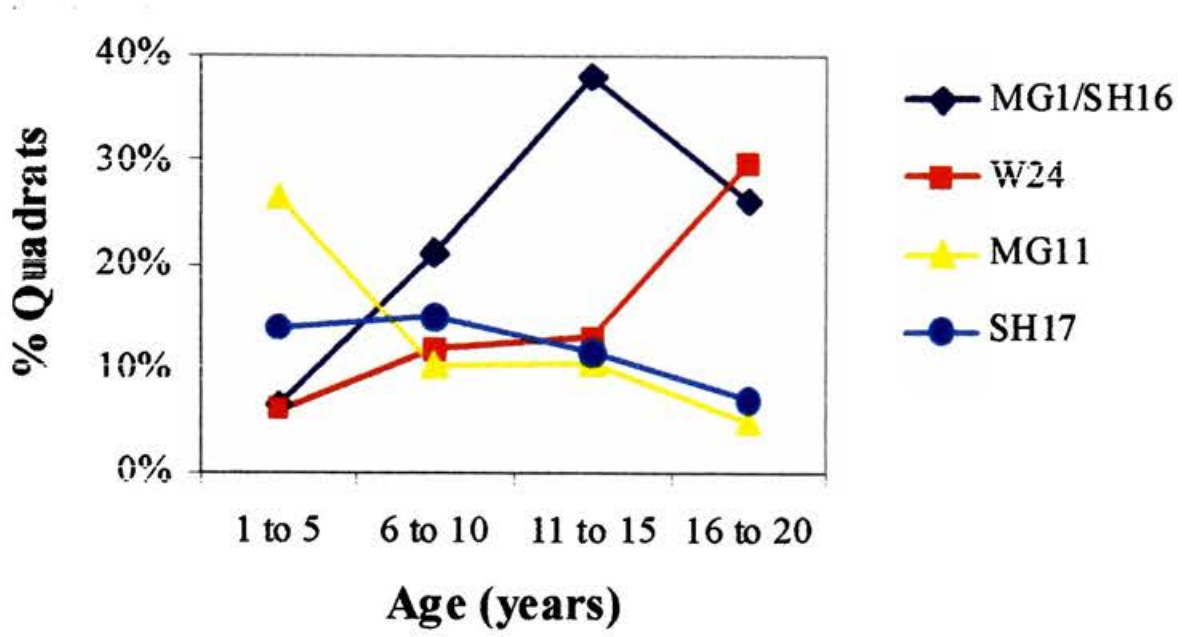
The graph also suggests that at around the 13-year mark, the abundance of MG1 peaks and begins to decline, coinciding with a rise in the abundance of understorey scrub (W24). This reduction in the dominant grass sward is associated with a recovery by several of the open vegetation communities (Fig 2.6b), most clearly by OV23 *Lolium perenne-Dactylis glomerata.* All of the commonly occurring communities of Shepherd's types (Fig 2.6c), principally tall herb and transition grassland, show steady decline after the 10-year mark.

Effect of substrate on the successional process

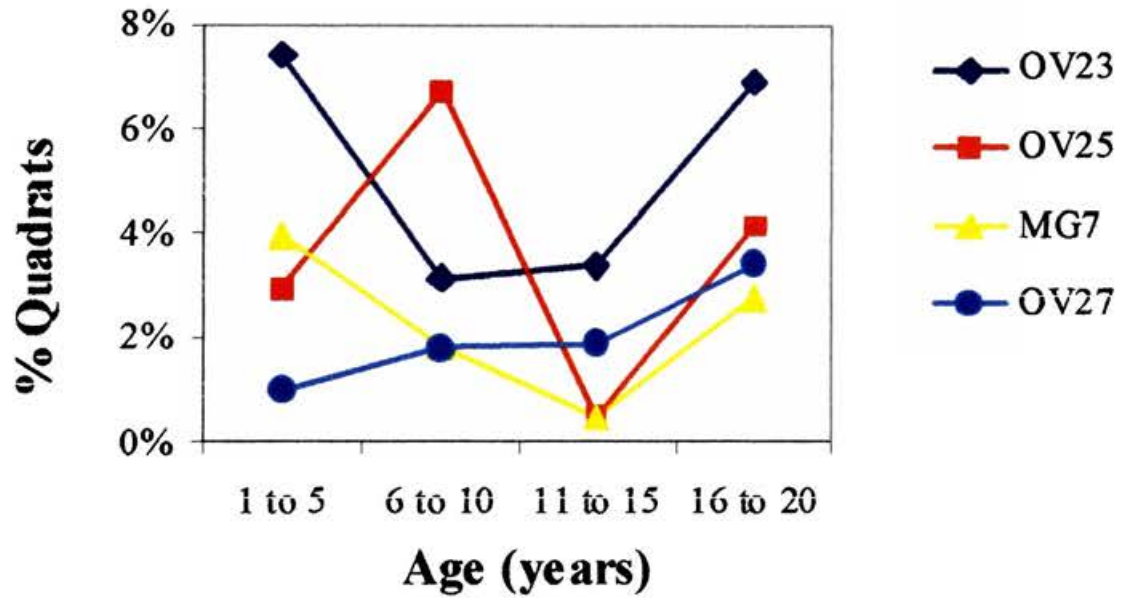
Figures 2.7 a-d display the frequency of communities found on the four most common substrate types surveyed (top soil, clay infill, concrete/broken tarmac and brick rubble). Each graph separates data from sites less than 10 years old from those that are 10 years old or above. Communities are ordered on the x-axis according to the difference in their frequency between the two time periods.

The results indicate that the speed of successional transition and, consequently community convergence to MG1, is fastest on topsoil (Figure 2.7a), where over 60% of quadrats are of W24 or MG1 type beyond the 10-year mark. On clay, (Figure 2.7b) the transition to MG1 *Arrhenatherum elatioris* is quite clear (MG1 accounting for 40% of quadrats at sites over 10 years old), but the development of understorey W24 *Rubus fruticosus-Holcus lanatus* understorey is apparently retarded by the substrate. In these circumstances, the MG11 *Festuca rubra-Agrostis stolonifera-Potentilla anserina* and SH17 *Holcus lanatus* transition continue to retain much of their importance with age. By contrast, on the sites over broken concrete, it is apparent that the understory community frequently becomes established earlier than MG1 *Arrhenatherum elatioris*, which reaches dominance much later (Figure 2.7c). Sites over brick rubble show a similar, but faster, pattern of transition than those over concrete, with woody vegetation (W24) initially more successful than the *Arrhenatheretum* (MG1) (Figure 2.7d).

Figure 2.6: *Changes over time in the frequency of 12 commonly occurring vegetation types of derelict land*



(b) Open vegetation



(c) Tall herb

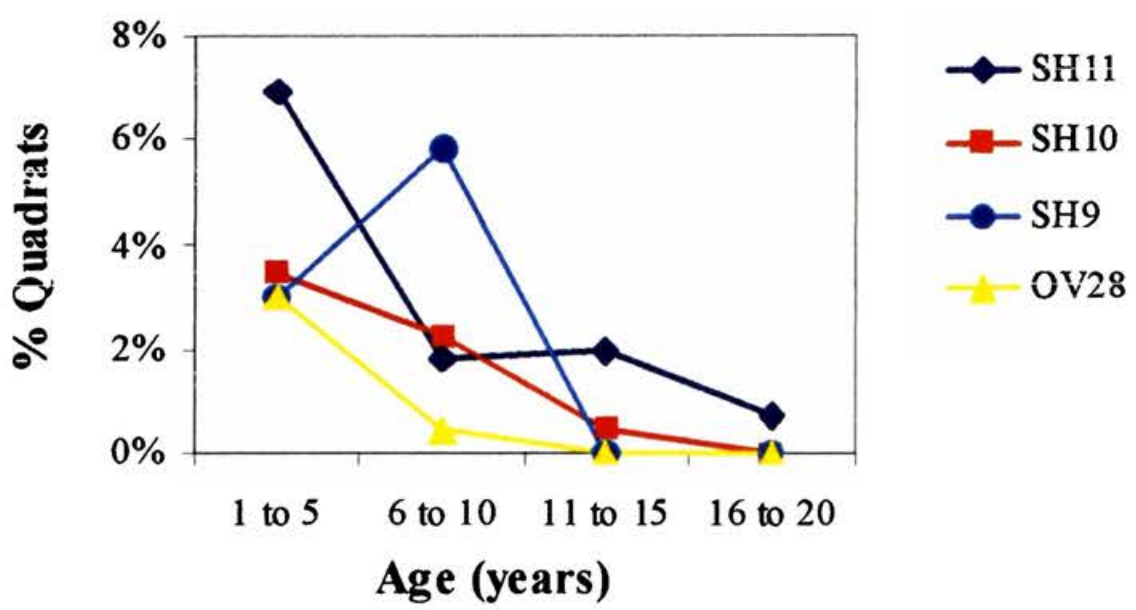
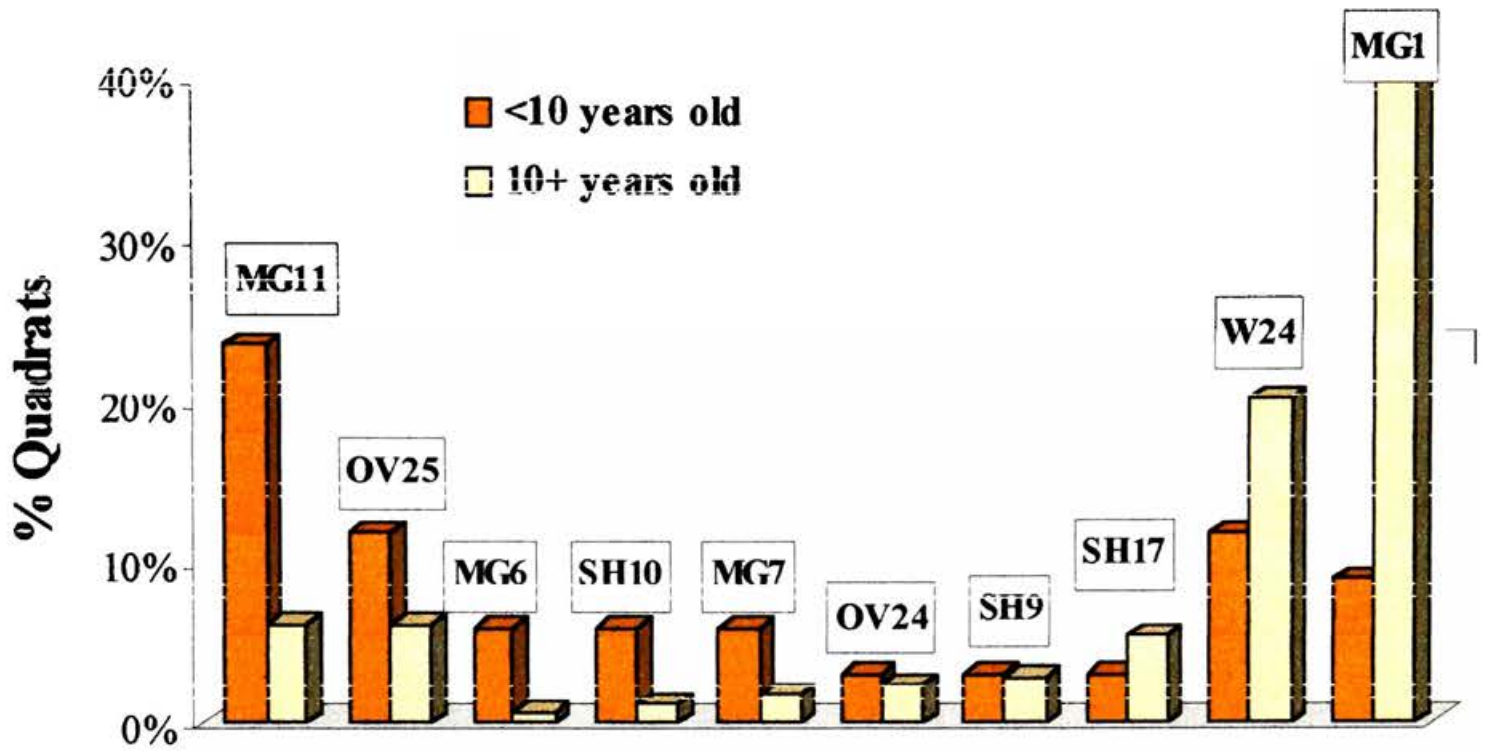


Figure 2.7: *Relative frequency of urban plant communities over different substrates.*

a) Topsoil



b) Clay infill

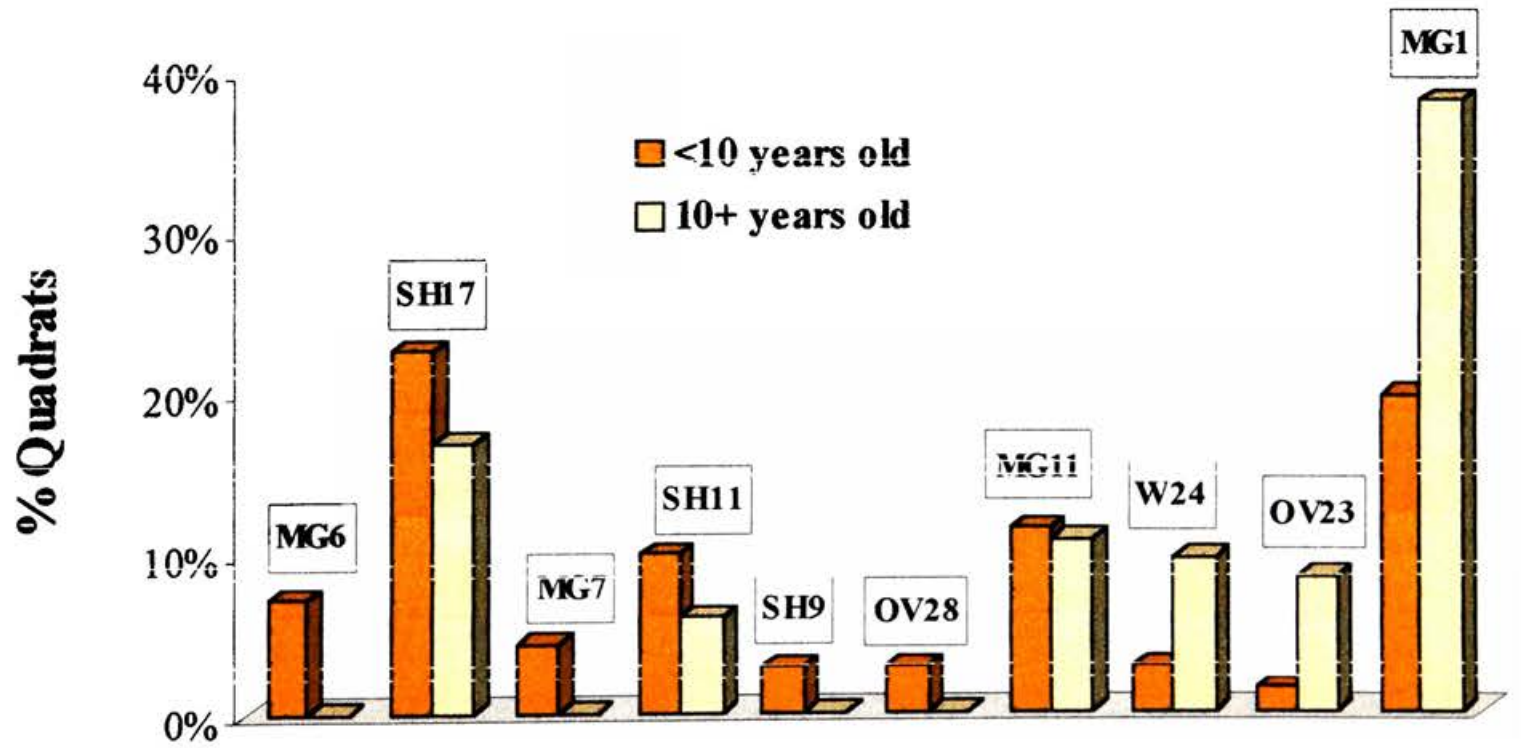
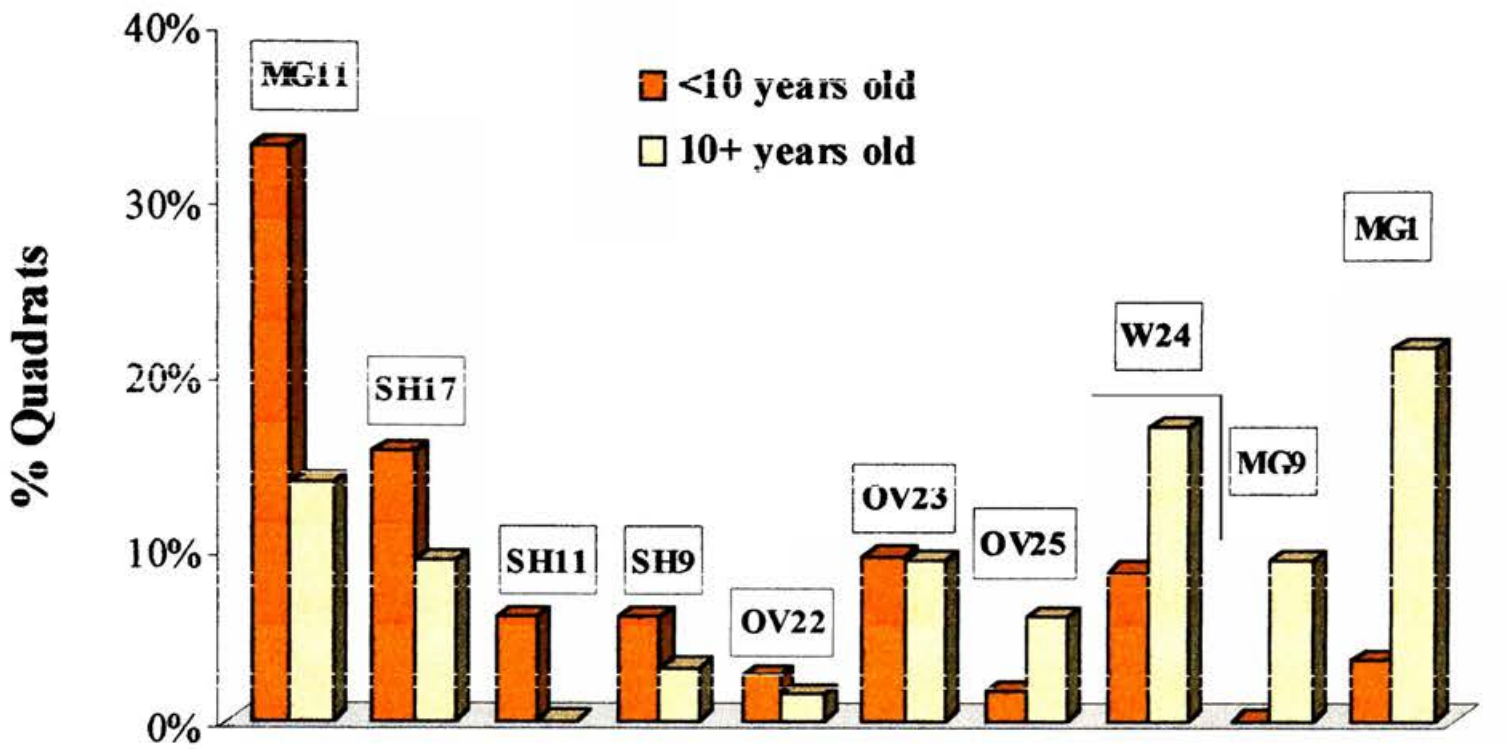
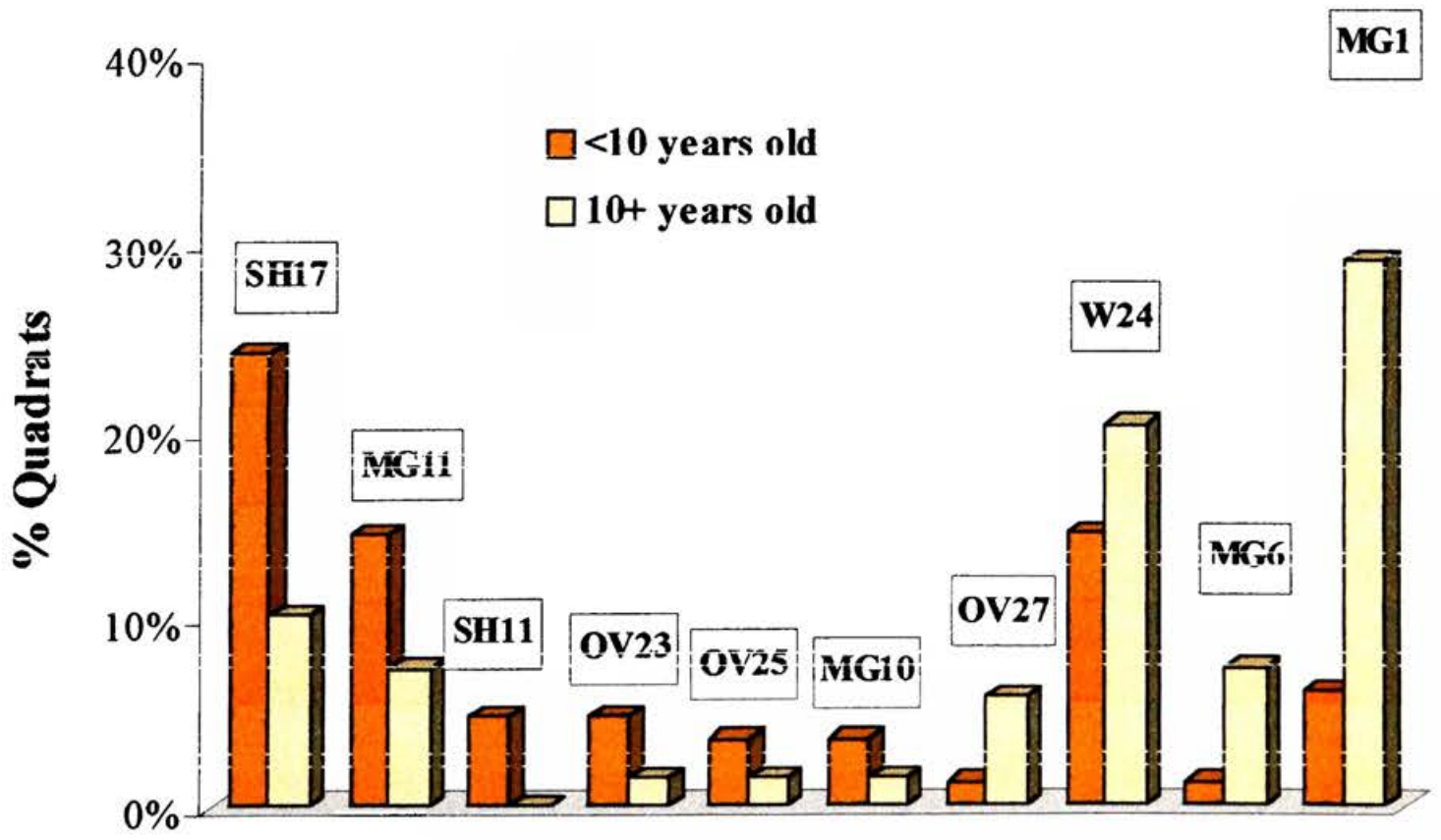


***Figure 2.7** (continued)*

c) Concrete / Broken tarmac



d) Brick rubble



2.5 RESULTS (3): A REVIEW OF COMMUNITIES FOUND ON DERELICT LAND SITES IN THE WEST MIDLANDS

The most frequently identified communities across the 50 sites surveyed in this study are shown in Table 2.5. These communities are reviewed below, highlighting their distribution in the survey in relation to site age and substrate type. Also identified are regularly occurring sub-communities and component species distinctive by their absence or repeated presence when compared to the association tables for NVC and Shepherd's types (see Appendix III).

Pioneer communities

Few previously described pioneer communities were found with any regularity in this study. This may be due in part to the fact that few very young sites were sampled in this study but may also reflect the difficulties with matching younger quadrats (see also Figure 2.3 and Table 2.4b).

Within the alliance Sisymbrium officinalis Shepherd recorded **SH6** *Senecio squalidus-Conyza canadensis* as the most common pioneer community, which is very similar to Gilbert's (1989) description of a *Senecio squalidus* colonization stage on bare surfaces. However in this survey, **SH6** was matched to only 1% of quadrats and was frequent at only one site. The rarity of Shepherd's other pioneer communities in this alliance can be largely attributed to habitat and location. **SH1** *Hordeetum murinii* is most typical of road edges, the base of walls and car parks, rather than large patches of derelict vegetation. **SH3** *Sisymbrium officinalis-Tripleurospermum inodorum* was noted by Shepherd as being frequent at the edges of sites on mounds and heaps and consequently perhaps has been not greatly recorded here. **SH2** *Sysimbrium loeseli* has only previously been described from Nottingham city centre which explains its non-appearance in this study.

The rarity of communities within the alliance Thero-Airion also be largely explained by habitat. Both the **SH14** *Hieracium pilosella* and **SH15** *Vulpia myuros* communities are almost exclusively associated with cinder track on disused railway, a habitat sampled on a part of one small site only in this survey.

Within the alliance Sambuco-Salicion capreae, the **SH12** *Epilobio-Salicetum caprea* pioneer

Table 2.5: *Most regularly occurring associations in derelict land field study (quadrat and site totals for communities include sub community totals where applicable)*

ID	COMMUNITY (SUBCOMMUNITY)	ALL TABLEFIT OUTPUT DATA		ONLY QUADS WITH GOF > 50%		INTER-QUARTILE AGE RANGE
		NO. QUADS	NO. SITES	NO. QUADS	NO. SITES	
SH6	*Senecio squalidus-Conyza canadensis*	11	7	2	2	n/a
SH9	*Artemisia vulgaris-Urtica dioica*	31	23	18	10	5.0-10.0
SH10	*Rumex obtusifolius* community	18	20	13	8	4.3-8.0
SH11	*Melitotetum albae-officinalis*	26	9	23	7	4.0-8.0
SH12	*Epilibio-Salicetum caprea*	14	8	5	5	6.5-11.0
SH17	*Holcus lanatus* transition community	114	34	94	28	5.0-14.0
MC9	*Festuca rubra-Holcus lanatus* grassland	25	13	20	11	8.0-18.9
MG1	*Arrhenatheretum elatioris* grassland	192	35	177	34	10.0-15.0
MG1a	*(Festuca rubra)*	77	22	66	19	n/a
MG5	*Cynosurus cristatus-Centaurea nigra* grassland	12	5	9	3	15.0-17.5
MG6	*Lolium perenne-Cynosurus* cristatus grassland	17	9	16	9	3.0-15.0
MG7	*Lolium perenne* leys	24	18	17	15	4.0-16.5
MG9	*Holcus lanatus-Deschampsia cespitosa* grassland	12	4	9	4	10.0-11.0
MG11	*Festuca rubra-Agrostis stolonifera-Potentilla anserina*	123	32	105	30	4.0-11.0
MG11a	*Lolium perenne*	108	31	95	29	n/a
OV22	*Poa annua-Taraxacum officinale*	11	7	4	4	n/a
OV23	*Lolium perenne-Dactylis glomerata*	53	28	39	21	4.0-17.0
OV23c	*Plantago major-Trifolium repens*	10	7	9	6	n/a
OV23d	*A. elatius-Melilotus lupulina*	20	15	15	12	n/a
OV24	*Urtica dioica-Galium aparine*	9	7	7	5	6.0-20.0
OV25	*Urtica dioica-Cirsium arvense*	36	22	28	17	6.3-13.8
OV25a	*Holcus lanatus-Poa annua*	13	10	7	7	n/a
OV26	*Epilobium hirsutum* Tall herb community	18	5	4	3	n/a
OV27	*Chamerion angustifolium*	21	13	15	12	10.0-17.0
OV27b	*(Urtica dioica-Cirsium arvense)*	17	13	14	12	n/a
W24	*Rubus fruticosus-Holcus lanatus* scrub under canopy	130	37	108	33	8.0-20.0
W24a	*(Cirsium arvense-Cirsium vulgare)*	58	26	50	24	n/a
W24b	*(A. elatius-H. sphondyllium)*	10	8	7	6	n/a

scrub community of xerophytic conditions that Shepherd found only in the inner city region of Birmingham was also recorded only occasionally (1.5%) in this study.

In addition, within the alliance Fumario-Euphorbion, the **SH7** *Fumaria officinalis-Veronica persica* community, and the closely equivalent **OV7** *Veronica persica-polita*, were barely recorded here but Shepherd himself observed this community to be rare in cities, except on freshly laid topsoil in landscaped flowerbeds. The most familiar NVC community in this alliance (**OV13** *Stellaria media-Capsella bursa-pastoris*) is typically found on fertile soils where regular cultivation effectively prevents any succession. It is expected that this community might be found on analogous habitats in cities, close to roadsides or paths, but it was not representative of colonising vegetation on actual brownfield sites.

Tall herb phase vegetation

SH11 *Melitotetum albae-officinalis* within the alliance Dauco-Melilotion was the only one of Shepherd's five distinct communities that was found frequently in the survey. It is assigned here to the tall herb phase because the dominant species and associate herb species, e.g. *Artemisia vulgaris*, contribute significantly to a high total biomass. However its position within a succession is not clear-cut because it arises to prominence as early as some pioneer stands. SH11 was sometimes the most abundant community on sites less than 5 years old particularly over clay, although it was also found on brick rubble and urban soil at lower frequencies. Shepherd also recorded this community most often on recently disturbed sites and found it especially amongst brick rubble and abandoned clay pits.

The matching between the survey quadrats and Shepherd's **SH11** association tables was very high, although *Melilotus alba* was found at considerably lower frequency in this study, and four species regarded in his study as frequent (*Elytrigia repens*, *Sonchus asper*, *Artemisia absinthium* and, in particular, *Daucus carota)* were not found at any of the survey quadrats. *Taraxacum officinale* was also found in this study with quite high frequency (25%) and was not identified in the original association.

Within the alliance Galio-Alliarion the **OV25** *Urtica dioica-Cirsium arvense* was regularly present on disturbed topsoil sites, most frequently on younger sites but also after suppression of the dominant grassland MG1 sward. The distribution pattern fits expected trends of the NVC categorisation that it is a community typical of nutrient-rich disturbed sites on loamy

soils. Within rural situations, **OV25** also often occurs where previous grassland stands have been suppressed, notably over dung pats or molehills. Flytipping can frequently lead to suppression of the dominant grass sward on derelict sites, resulting primarily in tall herb communities. The *Holcus lanatus-Poa annua* sub-community, which is the most frequent here, is also described as being characteristic of dumped soil on waste ground. The occasional presence of *Buddleja davidii* and *Chamerion angustifolium*, not reported within this association, gave a more urban element to this vegetation type in this study. The **OV24** *Urtica dioica-Galium aparine* community, which is also found within the Galio-Alliarion, was also recorded at several sites.

In the alliance Arction lappae only the **SH9** *Artemisia vulgaris-Urtica dioica* was widespread across the survey sites. This community was found irregularly on younger sites across a range of substrates. It is known to be characteristic of mounds and verges at the edges of sites which explains it's widespread, but low frequency, distribution. The survey quadrats showed close matching to Shepherd's SH9 association table, although *Urtica dioica* and *Anisantha sterilis* were found at considerably lower frequencies. *Lotus corniculatus* and *Epilobium ciliatum* were the only frequently found species here that Shepherd did not record in his Aufnahmen.

Another of Shepherd's communities, **SH10** *Rumex obtusifolius,* within the alliance Rumicion obtusifolii, was found infrequently but across a range of substrates. As with SH9, and as with all the regularly occurring communities described by Shepherd, it was most abundant at younger sites. SH10 was not recorded on sites beyond 14 years of age. It is also similar to SH9 in being primarily associated with mounds and verges and consequently was also widespread but never abundant at any particular site. *Urtica dioica* was found in much lower frequencies in this survey and *Arctium minus* and *Ballota nigra* were completely absent. In addition to these differences, several species not recorded by Shepherd were frequently found in this association, including *Taraxacum officnale* (43% of quadrats), *Sonchus oleraceous* (15%) and *Crepis vesicaria* (15%).

The major urban association within the alliance Carici piluliferae-Epilobion angustifolii is the **OV27** *Chamerion (Epilobium) angustifolium.* This was an infrequent but widespread community across the survey found on all substrates and age categories. Significantly it often

appeared as distinct patches on sites with little zonation, suggesting that very localised disturbance events had led to its development. The community is strongly associated with fire in many of its preferred non-urban situations, such as heathland or railway embankments, and it seems likely that the vast majority of patches of this community on wastelands can be attributed to accidental or deliberate fires. The high dispersal ability of *Chamerion angustifolium* seeds and the high productivity of the plant (Salisbury 1964) ensures that the absence of a proximate source is generally not an obstacle to colonisation of these burnt patches. The *Urtica dioica-Cirsium arvense* sub-community, which was most frequent here, is the type most associated with anthropogenic sites (Rodwell ed. 1999).

Trampled vegetation

The most commonly recorded vegetation types associated with trampling in this study belonged to the alliance Lolio-Plantaginion. **OV23** *Lolium perenne-Dactylis glomerata* was well distributed across the survey quadrats. It was most frequent in younger sites, particularly those where thicker grassland swards were slower to become dominant such as over concrete and clay sites. This corroborates NVC observations that this community is closely associated with paths and sites subjected to heavy trampling, essentially factors which will impede succession. In this survey however, this community was also frequent on older sites where woodland understorey **W24** replaces the thick grassland swards of **MG1**. Within the **OV23** association, both sub-communities *Arrhenatherum elatius-Medicago lupulina* and *Plantago major-Trifolium repens* were well represented. Both are typified by a high presence of *Holcus lanatus*. The high frequency of *Rubus fruticosus* in the quadrats of this survey and the occasional presence of woody species *Acer pseudoplatanus* and *Betula pendula* suggests some quadrats matching to **OV23** were in transition to an understorey community (**W24**).

Also within the Lolio-Plantaginion and recorded across a range of substrates and site ages in this survey were **MG7** *Lolium perenne* leys and related grasslands. This vegetation reached most prominence on younger sites over clay. This link with poorly drained sites reflects its rural distribution where this vegetation is often found on land in river valleys which is frequently under water for much of the year. The frequency of tall herbs *Rumex obtusifolius*, *Artemisia vulgaris* and *Sonchus asper* in association with this vegetation point to the

likelihood that the stands sampled were either fragmentary or in transition to typical tall herb communities.

The other regularly recorded trampled vegetation types were also within the alliance Lolio-Plantaginion. The most common of these were **OV21** *Poa annua-Plantago major*, on heavily trampled bare places, and **OV22** *Poa annua-Taraxacum officinale* on more lightly trampled sites. In contrast, the trampled communities of the alliances Polygono-Chenopodion polyspermii and Polgonion avicularis were only sparsely distributed on the survey sites.

Open grass swards

The **MG11** *Festuca rubra-Agrostis stolonifera-Potentilla anserina* community within the alliance Elymo-Rumicion crispi was one associations matched most frequently by TABLEFIT. Notably, *Potentilla anserina* was absent from all 105 quadrats matched to this community. Indeed, although *P. anserina* is known to be variably present in this association, it was poorly recorded in the West Midlands in general. In its absence, the principal benefactors at the ground layer appear to be *Taraxacum officinale* agg. and *Plantago major. Potentilla reptans* shows no increase in prominence. While it is possible that the survey quadrats were wrongly assigned to **MG11**, it is perhaps more likely that this was an urban variant of the rural association, particularly since the habitat in which it was recorded was similar to those expected. Matches within this community were most frequent on the younger derelict sites surveyed. **MG11** reached maximum abundance over concrete and tarmac and was also frequent on urban topsoils, while its frequency on clay was relatively low. This supports the observation that free-draining, moist, soils are considered an important habitat requirement for this vegetation. Nearly all stands recorded as this type (90%) fitted most closely to the *Lolium perenne* sub-community. A number of highly dispersive tall or medium herb species, including *Epilobium hirsutum, Sonchus asper, Sonchus oleraceous* and *Artemisia vulgaris,* appeared quite frequently in what were probably stands in zonation with drier patches. Also recorded within this alliance was **OV28** *Agrostis stolonifera-Ranunculus repens* community, which may also be considered as a slightly trampling tolerant association. It was principally found in heavily waterlogged locations.

In the alliance Cynosurion cristati the **MG6** *Lolium perenne-Cynosurus cristatus* was recorded occasionally at younger sites, particularly over clay or brick rubble. Most of the

recorded quadrats of this type showed close similarity to MG6a, which is the typical sub-community where *Agrostis stolonifera*, *Ranunculus repens* and *Cirsium vulgare* show increased prevalence. The infrequency of this community across the study area makes it difficult to draw many conclusions regarding the degree of matching to the quadrats.

Grassland transition state

Shepherd identified the **SH17** *Holcus lanatus* transition community as occurring during succession between the pioneer ruderal vegetation types and the development of the *Arrhenatheretum elatioris*. Its transitional status was emphasised in this study by its much greater frequency (>20%) on the urban substrates which tend to slow succession (such as concrete and clay) as compared to disturbed soil (3%). In general SH17 appears on sites quite early (from 3 years onwards) and its frequency peaked at sites between 5-8years old (depending on substrate) before undergoing a slow decline. The survey quadrats matched this association very accurately, although *Artemisia absinthium* was not such an important component in this survey. *Crepis vesicaria* was the only species recorded with any frequency that was not found by Shepherd.

Coarse grassland swards and understorey phase

The most frequently recorded community in this study was the **MG1** *Arrhenatherum elatius* grassland / **SH16** *Arrhenatheretum elatioris* within Arrhenatherion elatioris alliance. It was invariably the dominant vegetation at sites after 7-12 years before being gradually replaced by understorey (**W24**). Succession to **MG1** was most impeded over concrete, while the duration of its dominance appeared to be prolonged during the slow transition associated with clay sites. The often very species-poor *Festuca rubra* sub-community was particularly well represented and consequently many of the herbs associated with more diverse *Arrhenethereta* were absent or found at lower cover than expected. The regular occurrence of *Crataegus monogyna* undoubtedly represented zonation to the woodier community types. The abundance of *Vicia hirsuta* found in this sub-community, a species not normally associated, was probably particularly indicative of urban mesotrophic grasslands.

In the Calthion palustris alliance, the **MG9** *Holcus lanatus-Deschampsia cespitosa* was not very common on the derelict sites surveyed, although where present it often formed extensive stands.

The only common scrub and understorey association recorded was **W24** *Rubus fruticosus-Holcus lanatus* within the alliance Rubion subatlanticum. This was abundantly recorded and was most typical of older sites on soil or brick rubble. It was also often frequent in many recently derelict sites, particularly in situations where grass swards could not develop quickly, notably over broken concrete or tarmac. In these situations, the typical woody species of this community may quickly colonise cracks and their roots may contribute to the breaking up of the substrate. Under these circumstances, the combination of a more penetrable substrate and a build-up of organic matter often provides the opportunity for later invasion and domination by MG1 grassland. Much of this vegetation appeared similar to the described *Cirsium arvense-Cirsium vulgare* sub-community. Additional tall weedy species not described for this community but found frequently in association here were *Rumex obtusifolius* and *Artemisia vulgaris,* while *Calystegia sepium* sometimes scrambled in between, most commonly over concrete. *Plantago lanceolata and Trifolium pratense* also regularly remained at the ground layer.

Plate 2.1: *Pioneer vegetation on recently denuded site. These highly variable communities were poorly described by existing classifications of urban flora.*

Plate 2.2: *A typical* Chamerion angustifolium *OV27 stand which developed over a fire patch and is being gradually overgrown by saplings of* Acer pseudoplatanus.

Plate 2.3: *a)* Melilotus officinalis *and b)* Melilotus alba *are constant species in the* Melitotetum officinalis *association recorded in the Midlands by Shepherd (1992).*

(a)

(b)

Plate 2.4: *Tall herb phase species (a) Dominant stand of* Cirsium arvense *with* Dactylis glomerata *also present; (b)* Artemisia absinthium*, a species partcularly indicative of the West Midlands.*

(a)

(b)

Plate 2.5: *Open grassy vegetation resembling MG6. Forbs shown include* Centaurea nigra, Trifolium pratense *and* Achillea millefolium. *Graminoids less clearly illustrated include* Cynosurus cristatus, Lolium perenne *and* Festuca rubra.

Plate 2.6: *Typical transition stage from MG1 rank grassland to W24 understorey.* Arrhenatherum elatius *is still abundant but* Rubus fruiticosus *and* Calystegia *spp. are becoming increasingly dominant.*

2.6 DISCUSSION

Utility of NVC Classification on urban vegetation

Shepherd's association tables showed closest similarity to the open vegetation and weed communities of the NVC where the vegetation type was also well distributed in arable situations. In contrast, Shepherd's associations that were found to have no apparent equivalent within the NVC were primarily urban specialist groupings, particularly the *Melitotetum officinale* (*Melitotetum albae-officinalis* ***SH11***) and the *Senecio squalidus-Conyza canadensis* (**SH6**) communities. Consequently the application of a solely NVC approach to classifying urban vegetation, particularly early successional stands, will overlook many of the communities which make cities so botanically distinct.

Classification of pioneer stands

The absence or rarity of good matches to pioneer classifications found in this study can be attributed to at least two major factors: the inherent difficulties with pioneer classification in fragmented landscapes, and the problem of sample size. By definition, pioneer habitats often feature large areas of bare ground and the distribution of plant species is often limited to patches surrounding their original point of establishment from seed deposition. Observations on pioneer communities of three years and younger by Gilbert (1989) identified that many species, including *Taraxacum officinale* agg. and *Trifolium repens*, form large monocultures or colonies around their initial point of arrival. In these very early stages of succession, the chance factors of arrival and establishment may be the key to success and primarily determine the species present on the site, their distribution and their relative abundance. Environmental factors appear to play a smaller role at this stage, perhaps because colonising species can exploit a broader niche than species of mature vegetation (Parrish & Bazzaz 1982). At the site level, this means that the species assemblage will be a mixture of a common regional pool of highly-dispersive species and a more site-specific pool of less well-dispersed species which arrive by virtue of their close proximity to the site or by chance events, such as tipping. This results in pioneer stands of vegetation that are often variable and consequently difficult to classify. From a sampling perspective, these bare and exceptionally patchy habitats

provide a considerable challenge. The $1m^{-2}$ quadrats that were used in this survey are therefore perhaps unlikely to provide much of an overview of the vegetation of a pioneer stand even in situations where the vegetation might appear homogenous when viewed over a wide area. While a site may appear heterogenous at a small scale it may often appear homogeneous at a large scale (Allen & Starr 1982, Allen & Hoekstra 1990). This is clearly demonstrated by the increase of good matches recorded as sites aged. Indeed, the patchiness of several of the large young sites surveyed in this study was such that even Shepherd's 5 x 5 metre sample size would be unlikely to provide an adequate overview of the constant species of pioneer stands. Sampling is also more of a problem on very unstable substrates such as sandy slopes, where bare patches are more frequent. The grouping of small quadrats of similar vegetation type as carried out together in this study (Appendix III), may overcome some of the difficulty associated with patchiness but is still problematic if the community limits are unclear. Nevertheless, the underlying results do suggest that existing classifications of pioneer communities in Britain, including those of Shepherd's study, are inadequate describe the diversity of these vegetation types in urban areas in the Midlands. The pronounced floristic variation attributed between different cities observed both within the Midlands (Shepherd 1992) and nationally (Gilbert 1989) further accentuates the limitations of classification on these habitats.

Disturbance events

While vegetation type was clearly shown to change with age, it is apparent from some results that deriving this variable from the date since site dereliction is not always most applicable. The regular presence of OV27 *Chamerion angustifolium* stands across all ages hints clearly at irregular disturbances which set back succession in localised situations. This community has primarily been recorded at between 3-6 years from continuous observations of urban successions (Gilbert 1989) before declining in the succeeding grassland phase. Consequently the stands of this vegetation found in the survey probably underwent a major disturbance, most typically fire, 3-6 years prior to sampling, within sites of much greater age. Small-scale fires are frequently lit on the sites by youths, particularly where access is unhindered and burnt patches were regularly observed within older sites during surveying. The occurrence of open pioneer vegetation may also be prolonged or reoccur within older sites in areas exposed

to disturbance or trampling, particularly beside paths. The persistence of pioneer species seeds in the soil (see also Chapter 4) for long periods means that there is a potential recruitment source to take advantage of newly occurring gaps. Tall herb communities, characterised by *Artemisia vulgaris, Urtica dioica, Rumex obtusifolius* and *Cirsium arvense,* persist much longer on gypsy banks at the edges of sites and mounds of earth within site due to greater instability and continued tipping. Without these disturbances it is likely that the observed decline in community diversity and the convergence towards the Arrhenetheretum grassland and the understorey scrub would be even more pronounced.

Convergent succession

The convergence of vegetation stands with age is predicted by several models of succession (Clements 1916, Odum 1969, Connell & Slatyer 1977) and has been demonstrated on old field successions (Egler 1954, Leps 1987), forests (Horn 1974, 1976), and pampas (Facelli *et al.* 1990). On highly artificial substrates, convergence has also been demonstrated for old mine sites (Hatton & West 1987) and abandoned gravel pits (Borgegard 1990). However divergence has previously been observed during succession (Mathew 1979; Pineda *et al.* 1981; Glenn-Lewin 1981). Inouye & Tilman (1988) and Christensen & Peet (1981) recorded both divergent and convergent succession in their plots during old field and forest successions respectively. Convergent succession can be explained according to the concept that random fluctuations at a site will be increasingly damped down over time by interspecific interactions of the biotic community (Margulef 1974). Divergent succession has been attributed to differential nutrient availability (Tilman 1985), water availability (e.g. Collins *et al.* 1987) and topography (Pineda *et al.* 1981). The lack of organic matter and the prevalence of coarse material on wasteland sites means that water holding capacity is often low (Bradshaw & Chadwick 1980), but across the range of substrata surveyed in this study it would have been expected to be highly variable. The soil fertility would also be expected to fluctuate between the derelict sites studied at the onset of succession due to the diversity of previous land uses. However the findings put forward here suggest instead that, at least at the community level, the nature of the later successional communities is often very similar across a range of sites and different substrates. There are certainly difficulties with achieving an accurate sampling scale at all stages of succession, something not considered by the majority of studies

demonstrating convergent succession. However, the indication of the results here is that the major influence of substrate and its associated environmental factors on derelict sites appears to be in determining the speed of the succession and not direction. On the other hand, observations did suggest that topography often creates distinctions between vegetation on older sites. Distinctive communities may be more likely to arise from succession on other substrates upon which environmental differences are more extreme. The much smaller number of quadrats sampled over xerophytic sandy deposits, ballast and clinker apparently demonstrated less predictable successions.

Implications for urban conservation

The clear suggestion from this study is that the highest diversity both within and between derelict sites occurs while they are still young. Although this diversity may be retained longer if the substrate is such as to slow the convergence of vegetation, it is clear that the overall trend poses a problem for conservation that will require imaginative solutions. The continued turnover of derelict plots of land remains the best means to effectively safeguard species richness. However demand for building space and rising land prices suggest that future turnover may be too rapid to enable colonisation of newly derelict land. Meanwhile, the preservation of existing sites would be futile in the long term without intensive and consequently costly management to set back successional processes.

2.7 Conclusions

Developing a classification of early pioneer communities for British cities is problematic due to:

a) High diversity between sites due to i) to differential recruitment pools determined by proximate seed sources and chance colonisation events, ii) to differences in the development of the pioneer community because of the substrate and other physical and chemical attributes of the site.

b) A high degree of patchiness within sites as initial colonisers proliferate taking advantage of the lack of competition and high nutrient levels.

c) Considerable floristic variation between urban flora of different cities due to differences in climate and historical exploitation of plants

The chronosequence of sites surveyed suggests that over a 20-year period, convergent succession occurs on most sites, leading to domination by firstly Arrhenatheretum grassland and then *Rubus fruticosus-Holcus lanatus* understorey. However patterns of succession can vary due to the following:

a) The speed of the succession is influenced by substrate and is particularly impeded on heavy clay soils.

b) Over broken concrete or tarmac, woodland understorey species may be important before the dominant grassland phase, perhaps playing a role which leads to increased build up of organic matter and increasing the penetrability of the substrate for subsequent colonisation by perennial grasses.

c) In circumstances where there is continual or sporadic disturbance by trampling, tipping or fire, both pioneer and tall herb phases of the succession will persist on older sites.

CHAPTER THREE

FUNCTIONAL CHANGES IN PLANT COMMUNITIES DURING SUCCESSION IN URBAN DERELICT SITES

SUMMARY

In order to make the most of today's global research environment there is a need to be able to understand, test and compare the latest findings from across the world. For ecologists, the vast array of biological taxa can provide an obstacle to interpreting the significance of work carried out in other countries. This obstacle can be removed through the use of consistently measured plant functional attributes.

This study investigated how the representation of certain key plant traits changed during vegetation succession in urban areas. Using trait data from a number of existing published and unpublished sources, the species found at 50 derelict sites in the West Midlands, UK were examined for differences. The sites represented a chronosequence of 2-20 years of age. By developing a functional type approach to this work, the aim was to identify regularly occurring combinations of traits and their relationship with both the successional gradient and the variable environment of derelict sites. Ellenberg values were used to compare the environmental fidelity of the functional types.

As expected, the major variation between the sites was explained by the successional gradient. Changes in seed longevity, plant height, lateral spread, leaf dry matter content, specific leaf area, life form and strategy along this gradient conformed to predictions from both models and previous research on succession. However, contrary to expectations, both the mode of dispersal and seed weight showed no significant changes. It is suggested that the fragmented nature of a city's landscape may alter the patterns of recruitment during regeneration on urban sites. This may account for patterns of change in trait representation that do not fit expected models of succession.

Eleven functional types were isolated using TWINSPAN and Detrended Correspondence Analysis. Environmental differentiation between functional types was higher for later successional groups of species. Increasing environmental specialisation along the successional gradient is compatible with the concept that speed of arrival will predominantly determine the initial composition of sites. Stochastic recruitment engendered by the heterogeneous surrounding landscape and the impact of humans is probably an important factor in the high species diversity across younger sites.

KEYWORDS: *Succession, Urban ecology, Plant traits, Functional types, Ellenberg values.*

3.1 INTRODUCTION

A functional approach to investigating succession on derelict land

The identification of traditional plant communities is only of limited utility to ecologists. Classification can be a useful tool for applied conservation as it enables site mapping and can be used when formulating management guidelines. However plant ecologists are much more interested in the causes of separation between groups of species, particularly along environmental gradients. Classification techniques work best when they are used to describe the communities of discrete homogenous stands of vegetation, but are less effective when used in transitional zones that hold the answers to many ecological questions. As Kent *et al.* (1997) point out, the user of classification methods must by definition confront the problematic issue of when the transitional zone of vegetation deserves recognition as a community in its own right.

A direct ordination of species can give a much better indication of their positioning in relation to their environment and enables visualisation of the range and ecological amplitude of a species. However, from a global viewpoint, the use of species names is often redundant as they can serve little purpose in comparative study when the flora of one continent is so different to that of another. The comparison of vegetation processes across such divides requires a different approach. This can be achieved by examining changes in shared morphological and physiological plant characters along an environmental gradient. Consistent patterns of the environmental associations of plant traits have been described for several

floras (Grime *et al.* 1988; Boutin & Keddy 1993; McIntyre *et al.* 1995; Diaz & Cabido 1997; Thompson *et al.* 1998).

By assembling plants according to similarities in these attributes it is possible to generate functional types, which may be a powerful tool for making ecological predictions. The increased focus on the functional classification approach in recent years is a result of its potential as a device for predicting the effects of climate change (Gitay & Noble 1994) and land use (Thompson 1994). A definitive selection of traits, and standardisation of their measurement, is the key to a co-ordinated international approach and numerous authors are currently engaged in debating how to achieve this (e.g. Gaudet & Keddy 1988; Leishman & Westoby 1992; Keddy 1992; Grime *et al.* 1996; Grime *et al.* 1997; Westoby 1998; Weiher *et al.* 1999) or indeed whether it is possible (Grubb 1998). Classification of functional types can be a comprehensive scheme for a regional or national flora (e.g. Grime 1979) or may be targeted at particular habitats or environmental gradients (e.g. Diaz & Cabido 1997)

Changes in plant traits associated with succession

A considerable amount is already known about how plant characteristics change along successional gradients. Most attention has been focused on seed attributes, life form and strategy which are discussed below, after a preliminary discussion of the distinction between primary vs. secondary succession.

(i) Primary vs. secondary succession:

Clement's (1916) definition of succession considered two types described as primary or secondary. In a primary sere, all new colonisers must arrive at a new site from outside, while in a secondary sere, many seeds persist in the substrate between disturbances and can act as an additional or indeed the major source for recolonisation (Grubb 1987). Within the urban landscape, successions occurring on anthropogenic substrates produced by very extensive quarrying or excavations may be viewed as essentially primary (Leisman 1957; Borgegård 1990; Rebele 1992), except where large quantities of seed might be included within deposited "capping" material. Meanwhile 'human-induced' secondary successions, as described by Grubb (1987), occur particularly on abandoned agricultural fields at the urban edge and derelict gardens or allotments. Some degree of ambiguity remains in certain cases of human-induced successions (van Andel 1993). For example, succession occurring on brick rubble

from derelict buildings (probably the most frequent substrate of urban derelict sites) could potentially represent either primary or secondary succession depending upon the extent to which the underlying soil is exposed by demolition. However previous research on the changes in plant traits during human-induced successions has shown that they are reasonably consistent irrespective of whether a primary or secondary sere is being studied (Prach *et al.* 1997).

(ii) Seed characteristics

The ability to retain a persistent seed bank in the soil is clearly an advantage on sites undergoing secondary successions. Numerous authors have reported this link between the ability to colonise and the ability to produce persistent seed banks (e.g. Bertillier & Aloia 1997; Prach *et al.* 1997), and a number of models have also demonstrated the benefit of seed persistence in less stable habitats generally (Venable & Brown 1988; Rees 1993; Cohen 1966; Bulmer 1984). In contrast, trees and other species typical of later successional states and more stable habitats do not generally form long-term seed banks (Howard & Ashton 1967; Marquis 1975; Conn *et al.* 1984).

From a biological perspective, the occurrence of new open derelict sites is unpredictable in both time and space and the window of opportunity for colonisation is short. If seeds are not already present in the soil, the ability to arrive at a newly denuded site as quickly as possible is essential. Consequently, many successful species on new sites are small-seeded and wind-dispersed (Clements 1916; Salisbury 1942, 1953; Fenner 1987; McIntyre *et al.* 1995). Small-seeded genera are generally well represented at all stages of successional sere (e.g. Werner & Platt 1976). In contrast, large seeds are selectively advantageous in closed communities associated with late succession, especially if light resources are low (Salisbury 1974).

In addition to small size, the seeds of many pioneer species may have morphological adaptations that enhance dispersal potential. On derelict sites, several members of the most common family of plants, the Asteraceae, have a calyx adapted into a plumed pappus to increase dispersal by wind. One potential quantitative measure of the relative wind-dispersal potential of seeds that accounts for both weight and seed adaptations is seed terminal velocity (Askew *et al.* 1997). Attributes such as barbs or fleshy seed coats that promote the potential for dispersal by animals have often been shown to be more closely associated with later

successional habitats (e.g. Hodgson & Grime 1988). Several studies have highlighted the importance of seed-dispersal by birds during regeneration (e.g. Bossema 1979; Darley-Hill & Johnson 1981; Johnson & Adkisson 1985). The key factor determining the extent of seed-deposition by birds appears to be the availability and type of roosting vegetation (McDonnell & Stiles 1983), explaining the low importance of birds as vectors on early successional sites.

(iii) Strategy

Expected changes in strategy during a vegetation succession are described by Grime's (1979) theoretical model of plant strategies and are determined by the productivity of the substrate. In theory, as a site ages, the representation of ruderal species will decline and that of stress-tolerant species will increase. On more productive materials, competitive strategists will tend to be better represented in the intermediate and later stages of the succession. However substrate is extremely variable between, and indeed often within, urban derelict patches. Consequently changes in strategy with succession are expected to be variable in this study both between and within sites.

(iv) Life form

The relationship between Raunkiaer's (1934) life forms and successional gradients has been widely studied. Phanerophytes and hemicryptophytes generally increase along a successional gradient, while therophytes are associated with early successional stages (Houssard *et al.* 1980; Burrows 1990; Glenn-Lewin *et al.* 1992). The link between the abundance of therophytes and intensity of disturbance on similar sites has also been shown (McIntyre *et al.* 1995). In their study of human-induced successions, Prach *et al.* (1997) demonstrated the increase in phanerophytes and the decrease in therophytes as succession progressed, and also that other life forms showed no significant changes.

Aims of this study

This work tests the hypothesis that the majority of variation in between plant assemblages found on derelict land is related to successional age. Secondly, the age of the site (years since dereliction) is used to predict the vegetation composition, and the effectiveness of this predictor is assessed. A further aim is to compare the changes in dominant plant characteristics during derelict urban successions to those observed during studies of

succession in natural and semi-natural habitats. Finally, it is intended to group species on the basis of their functional attributes in order to examine in more detail how environmental constraints may influence the progress of regeneration and determine the outcome of succession at different types of derelict site.

3.2 METHODS

Field surveys took place at 50 derelict land sites between June and September in 1998 and 1999 (Figure 2.1 and Appendix I). Sites were selected either from reconnaissance surveys in the region or from the current derelict land database of the local councils' planning teams. Walking transects were undertaken across the whole site at 5-metre intervals to draw up a complete list of higher plant species present. In addition, measures of abundance were obtained using the Braun Blanquet cover scale at 1056 quadrats ($1m^2$) chosen by stratified sampling at the 50 sites. Nomenclature is according to Stace (1997).

Site age (Appendix IV) was derived from the derelict land database of the local councils, aerial photography, local ordnance surveys maps, and by information received in response to a questionnaire sent out to 800 residents and institutions adjacent to sites (see Appendix V). The youngest site had been bulldozed 2 years prior to the study period and the oldest site had lain derelict for 20 years.

Data relating to plant characteristics were obtained from a number of existing published and unpublished sources (see Table 3.1). The selection of plant characters was based on previous studies highlighting their importance during succession (Noble & Slatyer 1980; Glenn-Lewin *et al.* 1992; Brown 1992; Prach *et al.* 1997) constrained by the availability of data. The C-S-R values from Grime's comprehensive functional classification of the British flora and Raunkiaer's (1934) life forms were also incorporated into the analyses.

The environmental indicator values used were based on Ellenberg's *Zeigerwerte* (Ellenberg, 1979, 1988; Ellenberg *et al.* 1991), recalculated to explain ecological ranges of species for Britain and Ireland (Hill *et al.* 2000). These values can give an indirect measure for environmental variables and have been used to infer environmental conditions during succession in continental European research (Prach *et al.* 1999).

Table 3.1: *Data sources for plant characteristics used for developing a functional classification of the flora on urban derelict sites.*

PLANT CHARACTER	SOURCE
Seed longevity index (0-1)	Thompson *et al.* (1997)
Log seed weight (mg)	Grime *et al.* (1981)
Seed dispersal vector (Animal/Wind/Both)	Grime *et al.* (1988)
Specific Leaf Area (sqrt)	Ken Thompson (unpublished) (see Appendix VI)
Dry matter (mg)	Ken Thompson (unpublished) (see Appendix VI)
Life strategy (C,S,R)	Grime *et al.* (1999)
Life form (Raunkiaer's scheme)	Clapham *et al.* (1952)
VA Mycorrhiza presence	Grime *et al.* (1988)
Lateral growth (1-5)	Grime *et al.* (1988)
Plant maximum height (cm)	Stace (1997)

3.3 DATA ANALYSIS

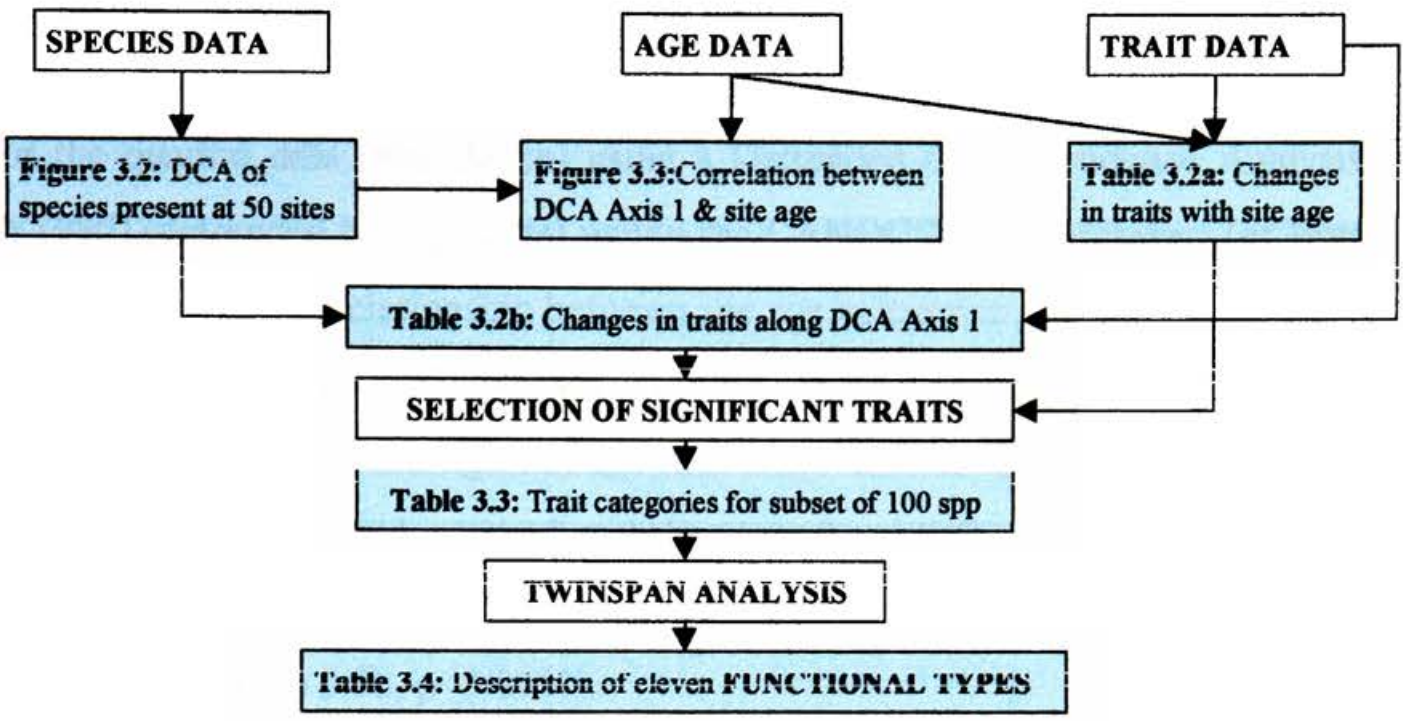
An outline of the analysis procedure described below is given in Figure 3.1a-c. The first section of results (Figure 3.1a) examines the changes in traits along the successional gradient. Traits that show significant changes with succession are then used to divide species into functional types.

In the second section of results (Figure 3.1b) the characteristic differences between functional types in terms of their traits are identified. Functional types are divided into groups relating to the pioneer, intermediate and later successional phases.

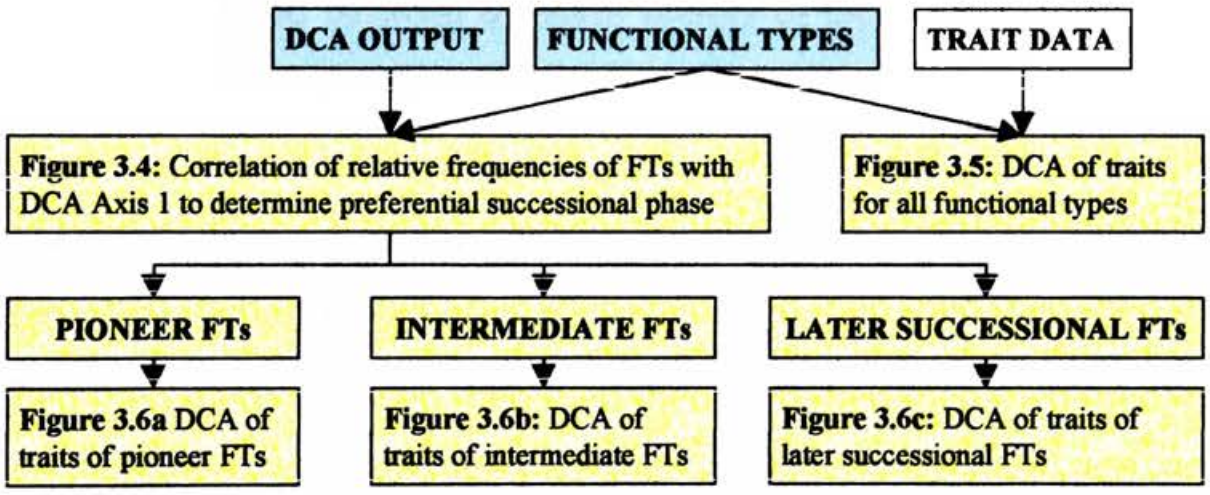
In the third section of results (Figure 3.1c), the environmental preferences of the functional types are examined using data on species Ellenberg values. This is used to investigate changes in environmental preferences over the course of succession.

Figure 3.1: *Flow diagram illustrating data analysis procedure*

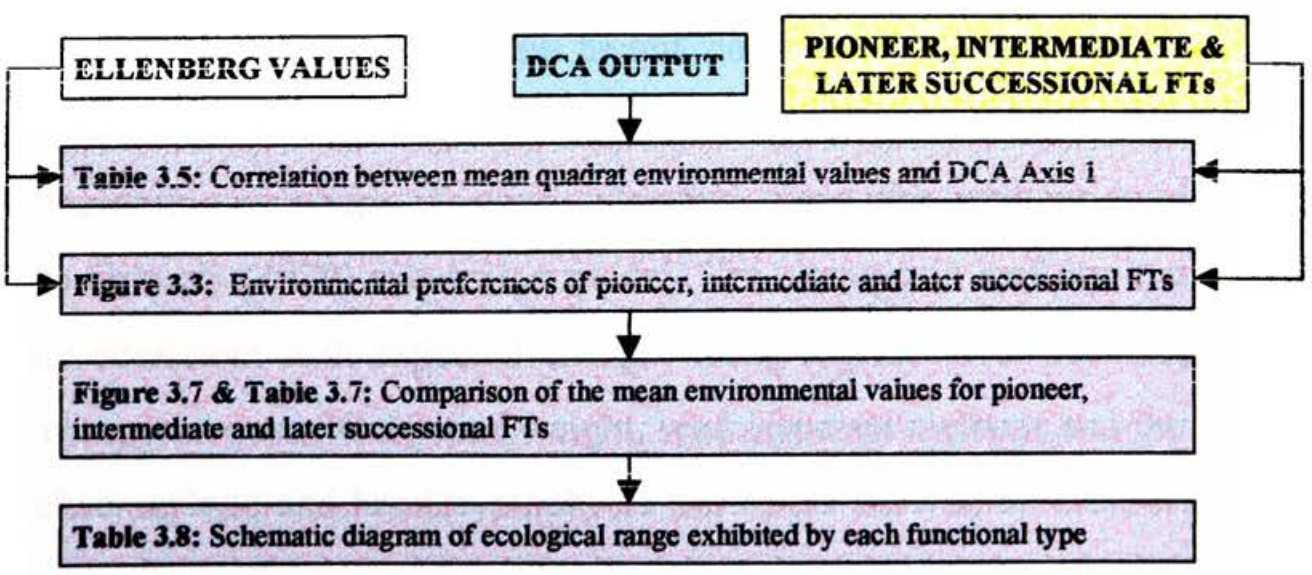
(a) Analysing changes in traits along the successional gradient and identification of functional types.



(b) Analysing the characteristic differences between the functional types



(c) Describing the environmental preferences of the eleven functional types and changes over succession.



3.4 RESULTS (1): CHANGES IN PLANT CHARACTERISTICS ALONG THE SUCCESSIONAL GRADIENT AND THE IDENTIFICATION OF FUNCTIONAL TYPES

Ordination of the 50 sites was carried out on both the site assemblages (presence/absence) and for the quadrat data (abundance) using a Detrended Correspondence Analysis (Hill & Gauch 1980) (detrended by segments) within the CANOCO v.4.0 package (Ter Braak 1997). Figure 3.2 illustrates the relationship between site age categories and axes 1 and 2 of the DCA for the species assemblages. There was an apparent relationship between age and the first axis. The correlation between age and the first DCA axis of the presence/absence data was tested by the Spearman Rank method and shown to be significant (Figure 3.3a, $p<0.001$). A significant relationship was also shown when this process was repeated for the quadrat abundance data (Figure 3.3b, $p<0.0001$).

Having established an age-related trend, the changes in the representation or values of the chosen traits were examined between different sectors of the successional gradient. Sectors were obtained in two ways. First by dividing the sites into three groups according to their age since dereliction (2-6 years, 7-13 years, 14+ years); and secondly by categorising sites into three groups of the same size according to their position along the first axis of DCA output. A one-way Analysis of Variance was used on each of the two sets of sectors to investigate changes in traits along the gradient. Tukey multiple comparison tests were used to test for differences between sectors. The results are exhibited in Tables 3.2a for age and Table 3.2b for DCA axis 1.

Both tables show similar expected patterns of change. A decrease in seed longevity, ruderals and therophytes and an increase in plant height, competitors, phanerophytes and mycorrhiza presence is consistent with findings in a previous successional study of plant traits (Prach *et al* 1997). Increasing numbers of stress tolerators along the gradient is also as expected according to Grime's (1979) succession model. The changes in leaf dry matter and specific leaf area are consistent with expected changes along a gradient of declining ruderality and increasing competitiveness. Only seed weight, seed dispersal methods and the participation of geophyte, chaemaphyte and hemicryptophytes life forms showed no clear relationship along

the DCA gradient. The absence of an observed significant trend for seed weight and method of dispersal was contrary to the expected findings.

The similarity between findings based on age and those based on the DCA axis 1 reflects the relationship between age and the first axis. However, significant changes in the traits are generally most clearly reflected along the DCA gradient. This suggests that alignment along the first DCA axis was a better indicator of successional status than age of site.

Analyses to identify functional types were conducted on a subset of the total species list. In order to focus on those species most responsible for the variation between sites, only those species found at more than 3 and less than 40 sites were incorporated within this subset. The most frequently recorded 100 species within this subset were then used. Plant trait values for each species in the subset were then translated into categories (see Table 3.3). Traits that showed no clear response in the earlier analysis were removed to prevent the possibility that they may confuse patterns described by further analysis. This refined data set of species and trait groupings was then analysed using TWINSPAN (Hill 1979) to identify groupings of species according to their functional similarity. Eleven functional types were identified, and these are described in Table 3.4.

Figure 3.2: *Axis 1 by axis 2 plot of a Detrended Correspondence Analysis of the species presence/absence data for 50 derelict sites. Sites identified by age band*

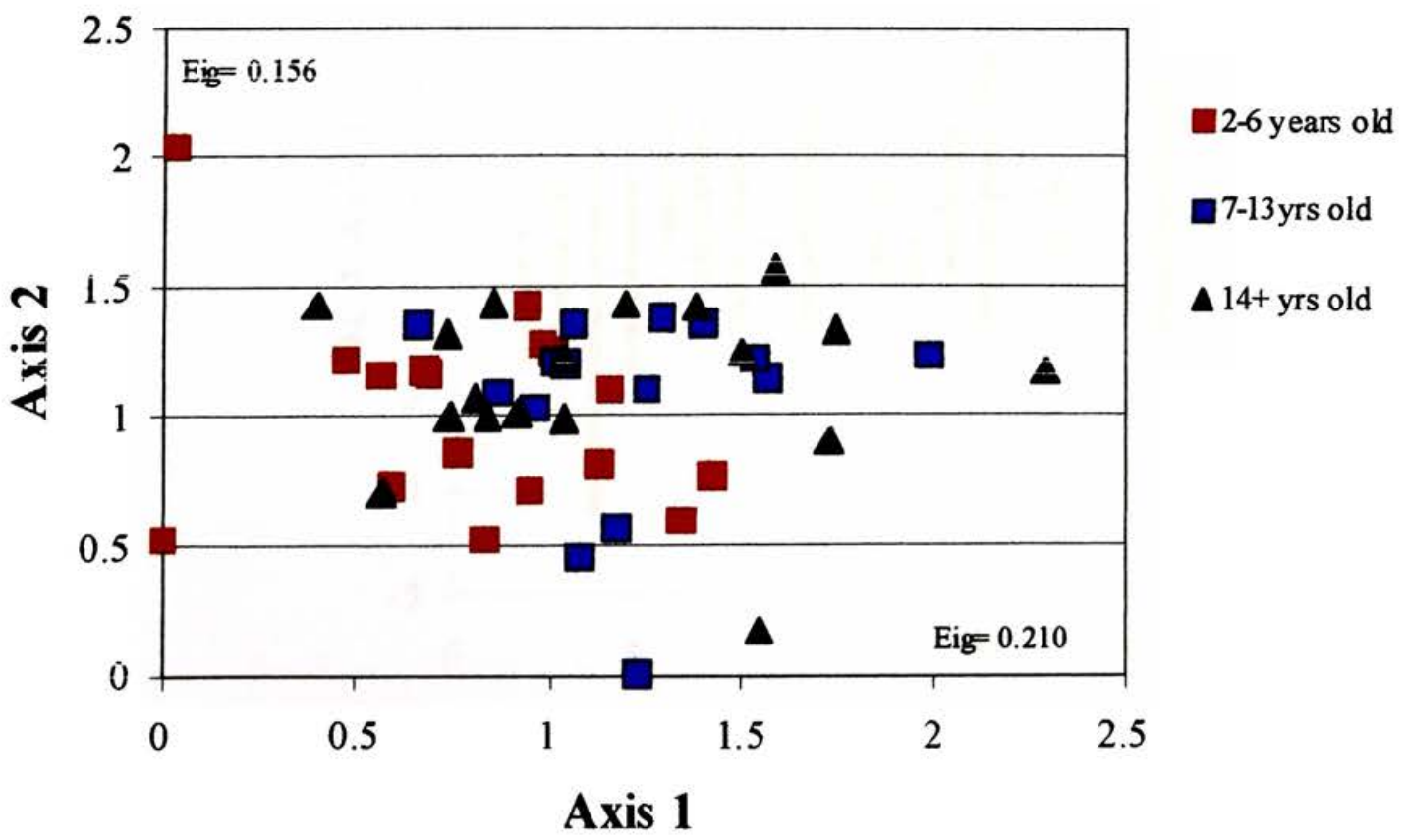
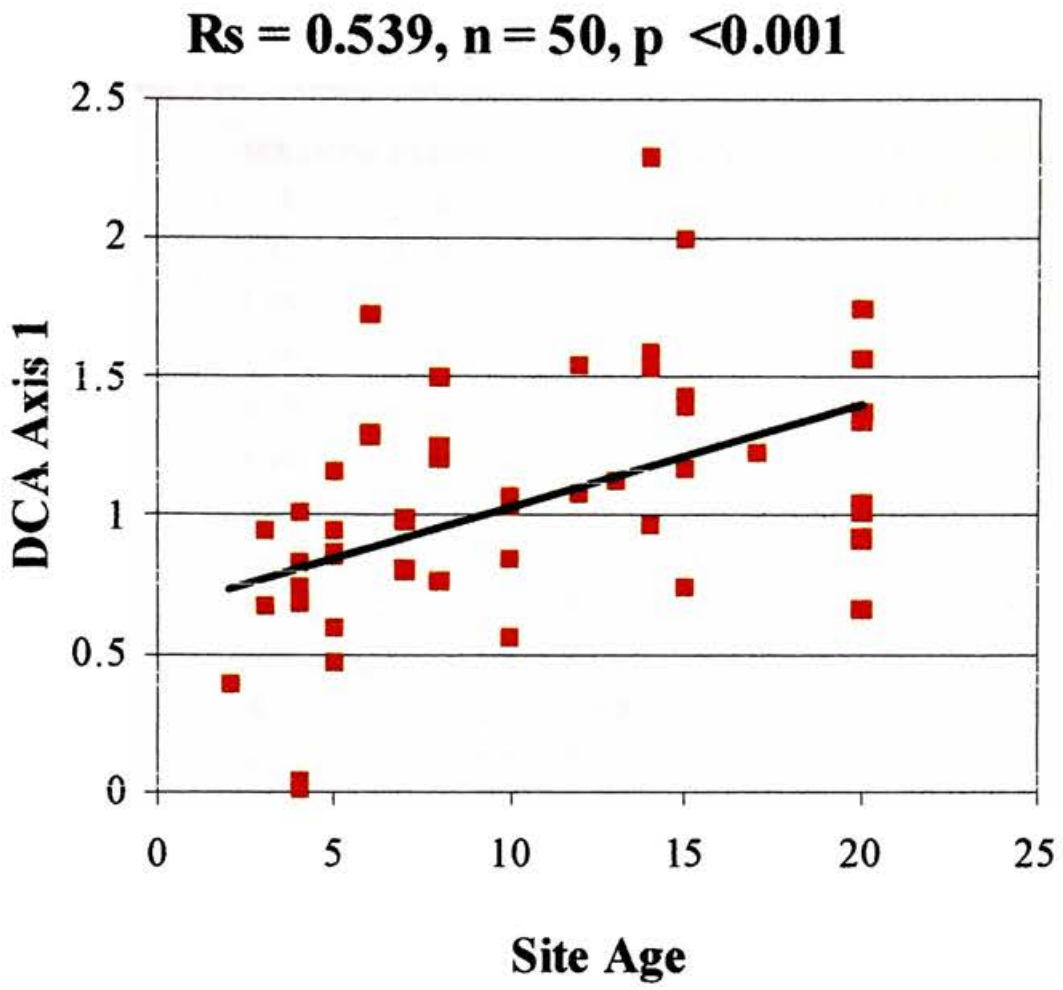


Figure 3.3: *Correlations between site age and axis 1 of the DCA output of (a) site presence/absence data and (b) quadrat abundance data*

(a)



(b)

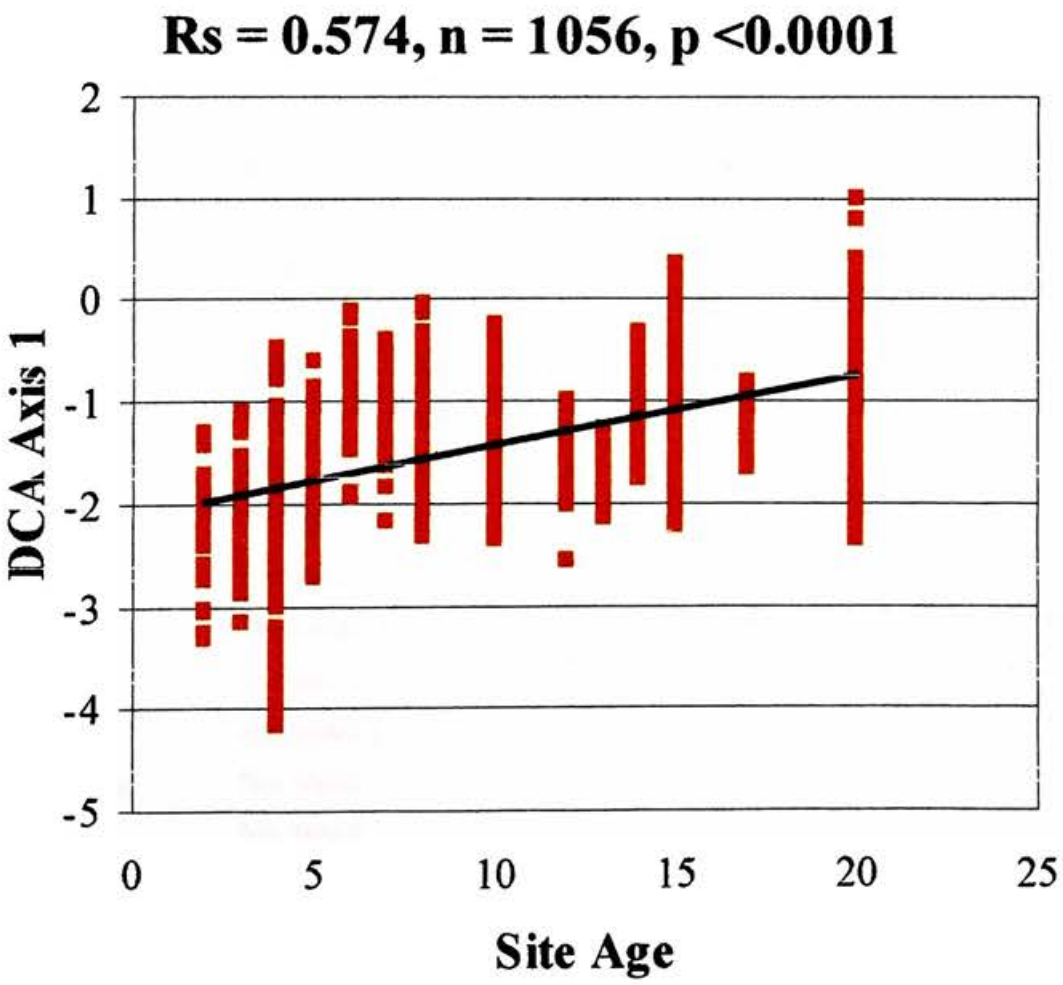


Table 3.2: *Differences in plant characters between groups of sites tested by One-way ANOVA and Tukey multiple comparison tests (- not significant, † p<0.1, * p<0.05, ** p<0.01, *** p<0.005, **** p<0.001).*

(a) Sites grouped by age: A: Pioneer sites (2-6 years old, n=17), B: Intermediate sites (7-12 years old, n=15), C: Later sites (14+ years old, n=18).

PLANT CHARACTER	MEAN OR TREND			ANOVA	TUKEY TESTS		
	A	B	C		A vs. B	A vs. C	B vs. C
Seed weight (mg)	2.33	2.30	2.48	-	-	-	-
Longevity index	1.49	1.47	1.34	****	-	****	-
Lateral growth potential	3.79	3.98	3.94	**	*	-	-
Leaf dry matter content	4.49	4.56	4.56	***	†	†	-
Surface Leaf area (sqrt)	4.96	4.90	4.88	†	-	†	-
Plant height (cm)	283.1	324.9	297.1	-	-	-	-
Participation of Competitors	Increasing →			†	*	-	-
Participation of Ruderals	Decreasing →			****	-	****	****
Participation of Stress tolerators	Increasing →			****	-	****	*
Participation of Phanerograms	← Decreasing →			**	*	-	*
Participation of Hemicryptophytes	Increasing →			*	-	*	*
Participation of Geophytes	No trend			-	-	-	-
Participation of Therophytes	Decreasing →			****	***	****	-
Participation of Chaemaphytes	No trend			-	-	-	-
Dispersal by wind	No trend			-	-	-	-
Dispersal by animal	No trend →			-	-	-	-
Presence of Mycorrhiza	Increasing			†	-	†	-

(b) Sites grouped by DCA axis 1 score: S1: Pioneer sites (DCA Axis 1 score 0.0-0.85, n=17), S2: Intermediate sites (0.86-1.15, n=15), S3: Later sites (1.16-2.30, n=18).

PLANT CHARACTER	MEAN OR TREND			ANOVA	TUKEY TESTS		
	S1	S2	S3		S1 vs S2	S1 vs. S3	S2 vs. S3
Seed weight (mg)	2.38	2.12	2.58	-	-	-	-
Longevity index	1.47	1.48	1.43	***	-	***	***
Lateral growth potential	3.80	3.80	4.03	****	-	****	****
Leaf dry matter content	4.48	4.48	4.60	***	-	***	***
Surface Leaf area (sqrt)	4.95	4.95	4.87	*	-	-	-
Plant height (cm)	256.20	261.20	351.50	****	-	*	****
Participation of Competitors	Increasing →			*	-	*	-
Participation of Ruderals	Decreasing →			****	-	****	****
Participation of Stress tolerators	Increasing →			****	-	****	****
Participation of Phanerograms	Increasing →			****	-	****	****
Participation of Hemicryptophytes	No trend			-	-	-	-
Participation of Geophytes	No trend			-	-	-	-
Participation of Therophytes	Decreasing →			****	-	****	****
Participation of Chaemaphytes	Increasing			-	-	-	-
Dispersal by wind	No trend			-	-	-	-
Dispersal by animal	No trend →			-	-	-	-
Presence of Mycorrhiza	Increasing			†	-	-	-

Table 3.3: *Categories used for TWINSPAN and DECORANA determination of functional types (see Table 3.1 earlier for source material)*

PLANT CHARACTER	CATEGORIES	VALUE
SEED LONGEVITY	LOW	0.00-0.20
	MID	0.20-0.65
	HIGH	0.65-1.0
PLANT HEIGHT	LOW	0-50cm
	MIDL	51-90cm
	MIDH	91-150cm
	HIGH	150cm+
MYCORRHIZA	Presence	Infection on more than 25% of records
LATERAL GROWTH	LOW	1-2
	MID	3-4
	HIGH	5-6
DRY MATTER	LOW	0.0-4.0
	MID	4.0-5.0
	HIGH	5.0+
SPECIFIC LEAF AREA	LOW	0.0-4.5
	MID	4.5-5.2
	HIGH	5.2+
PHANEROGRAMS	Presence	Y/N
THEROPHYTES	Presence	Y/N
COMPETITOR STRATEGY	HIGH	C, CR, SC, C/CR, C/SC, C/CSR
	MID	CR/CSR, SC/CSR, CSR
	LOW	All other strategies
RUDERAL STRATEGY	HIGH	R, CR, SR, R/CR, R/SR, R/CSR
	MID	CR/CSR, SR/CSR, CSR
	LOW	All other strategies
STRESS TOLERATOR STATEGY	HIGH	S, SR, SC, S/SC, S/SR, S/CSR
	MID	SC/CSR, SR/CSR, CSR
	LOW	All other strategies

Table 3.4: *Eleven functional types derived from TWINSPAN analysis of plant attributes for 100 species found on derelict land sites in the West Midlands.*

FUNCTIONAL TYPE	NO. OF SPECIES	DESCRIPTION	EXAMPLE SPECIES
FT1	9	Competitive ruderals Variable height Hemicryptophytes Medium to high seed longevity Medium to low leaf dry matter content Medium to high specific leaf area	*Ranunculus repens, Linaria vulgaris, Sonchus arvensis*
FT2	16	Ruderals Medium to low height Mostly therophytes Poor lateral growth Medium to high seed longevity Medium leaf dry matter Variable specific leaf area	*Trifolium dubium, Papaver rhoeas, Persicaria maculosa*
FT3	7	Ruderal or competitive ruderals Predominantly tall herbs Therophytes or Hemicryptophytes High seed longevity Mostly low leaf dry matter content Medium to high specific leaf area	*Rumex crispus, Sonchus oleraceaus, Sisymbrium officinale*
FT4	9	Predominantly ruderal strategists Low herbs Therophytes or Hemicryptophytes Poor to moderate lateral growth High seed longevity Low leaf dry matter Medium to high Specific leaf area	*Plantago major, Senecio vulgaris, Stellaria media*
FT5	9	Competitive ruderals Medium height herbs and grasses Hemicryptophytes or chamaephytes Moderate lateral growth Medium leaf dry matter content High seed longevity	*Poa trivialis, Tanacetum parthenium, Tragopogon pratense*
FT6	7	Strongly competitive Medium to high leaf dry matter content High lateral growth	*Calystegia sepium, Vicia cracca*

***Table 3.4** (continued)*

FUNCTIONAL TYPE	NO. OF SPECIES	DESCRIPTION	EXAMPLE SPECIES
FT7	5	Competitive ruderals Mostly tall herbs Hemicryptophytes or chamaephytes Moderate lateral growth Medium to high leaf dry matter content Variable seed longevity	*Artemisia absinthium, Armoracia rusticana, Artemisia vulgaris*
FT8	16	Strongly competitive and/or Stress tolerators Tall growing Mostly phanerograms High lateral growth High leaf dry matter content Low specific leaf area Low seed longevity	*Crataegus monogyna, Sorbus aucuparia, Sambucus nigra*
FT9	11	Predominantly CSR strategists Low or medium in height Hemicryptophytes Moderate lateral growth Medium seed longevity Medium or high specific leaf area	*Trifolium pratense, Hypochaeris radicata, Phleum pratense*
FT10	5	Stress tolerators Low herbs Medium leaf dry matter content Medium seed longevity	*Pilosella officinarum, Trifolium arvense*
FT11	6	Stress tolerators or CSR strategists Low seed longevity Medium low height Medium high leaf dry matter Medium low specific leaf area	*Anthyllis vulneraria, Cynosurus cristatus*
Total	100		

3.5 RESULTS (2): DEFINING THE CHARACTERISTIC DIFFERENCES BETWEEN THE FUNCTIONAL TYPES

The relative abundance of each functional type was calculated for each site as a proportion of the species present at each site. This value was then plotted against axis 1 of the DCA output for site assemblage in order to investigate changes in FT frequency during the course of succession (Figure 3.4). The number of species representing FTs 1-4 declined along this axis,indicating that their characteristic traits were most beneficial during the early stages of the succession. There was a significant increase in the component species of FTs 8-11 along the gradient indicating that their characteristic traits are beneficial in later succession. Species in FTs 5-6 showed no significant increase in either direction of the gradient and their collection of traits can be assumed to be either most advantageous at an intermediate stage of the successional seres studied, or to have no specific preference.

A DCA was then performed on the trait categories for the full set of 100 species in order to clarify the major functional separation between FTs associated with different age bands (Figure 3.5). This emphasises that the separation of functional types broadly reflects the significant changes in characters observed in Table 3.2a-b.

In order to investigate the much smaller differences between the FTs associated with the same successional phase, separate DCA's were performed on the species trait categories for FTs 1-4, FTs 5-7 and FTs 8-11 (Figure 3.6 a-c). Separation between pioneer types on the first two axes of the output was mostly determined by height, lateral growth, longevity and whether species were therophytes or hemicryptophytes. Species found in functional types dominant at older sites are divided according to whether they are competitors or stress tolerators, and whether they are phanerophytes or hemicryptophytes (and consequently also by height).

Figure 3.4: *Frequency of each functional type as a proportion of all the species present at each site, plotted against site DCA axis score (from Figure 3.2.)*

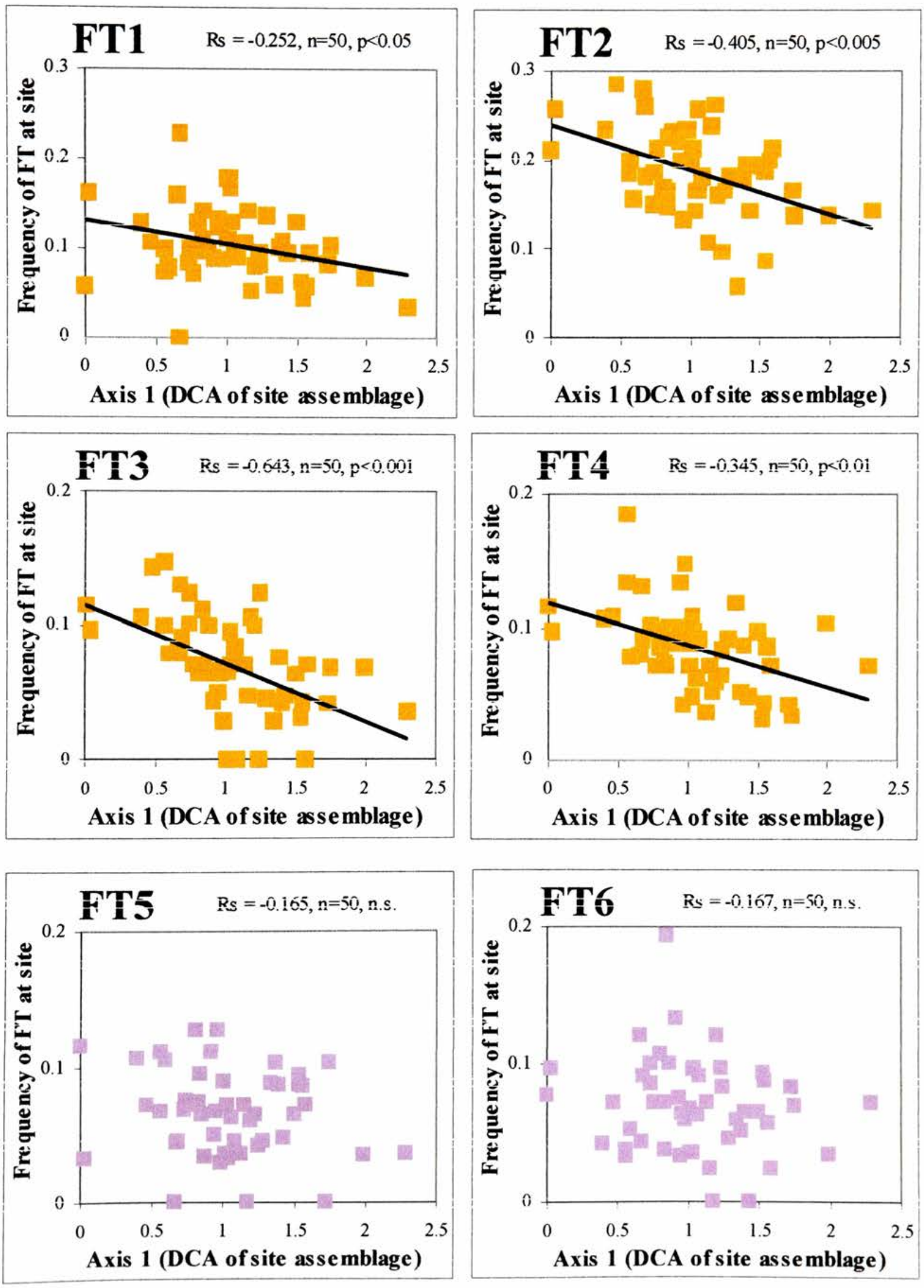
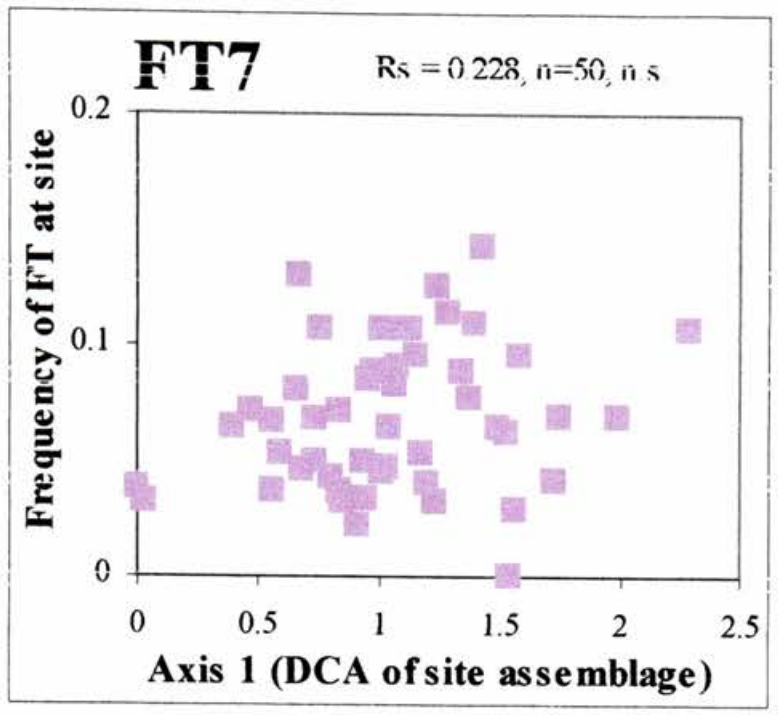
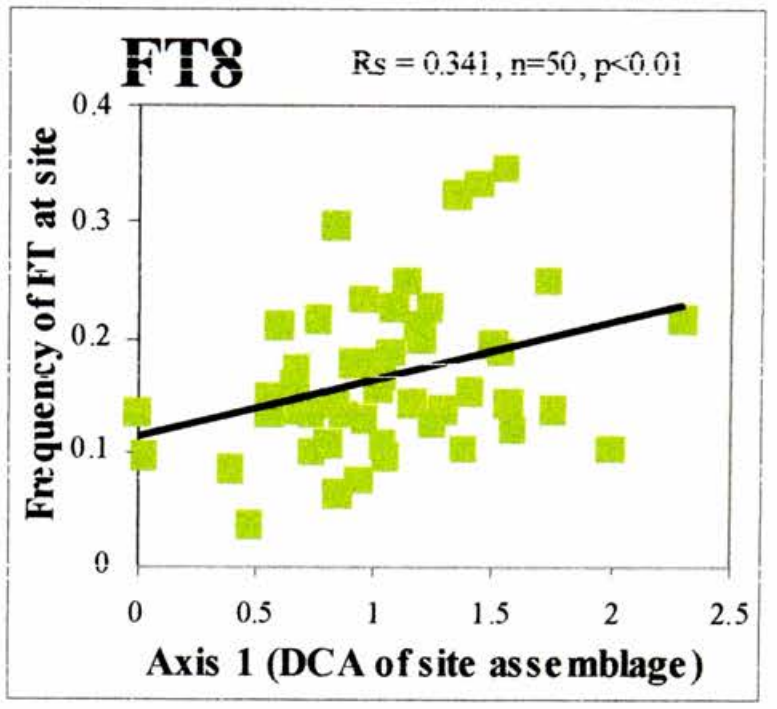
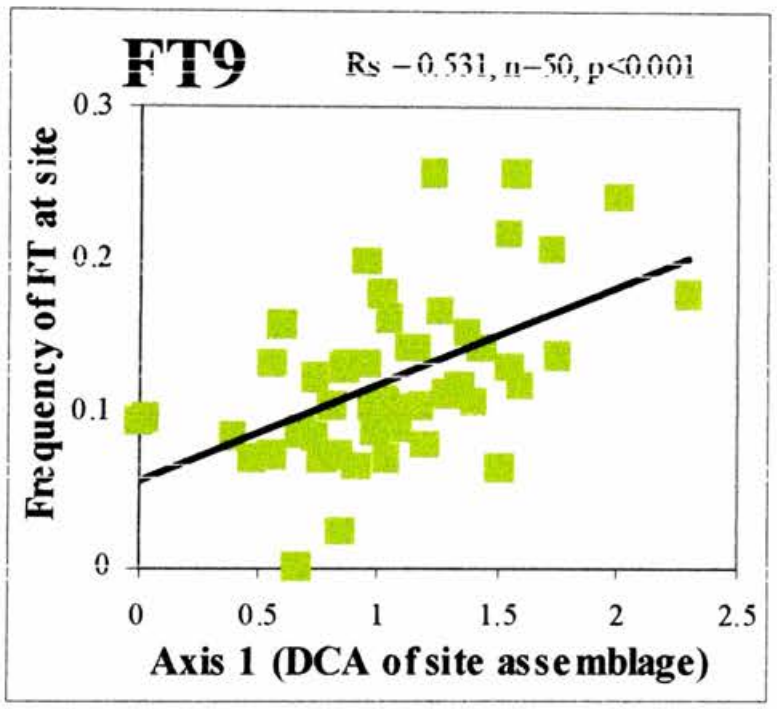
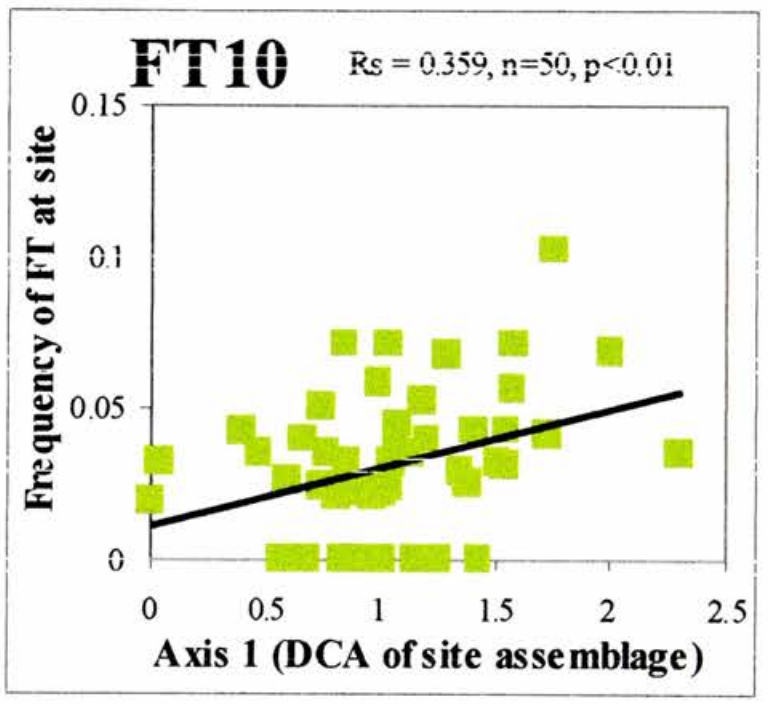


Figure 3.4: *(continued)*









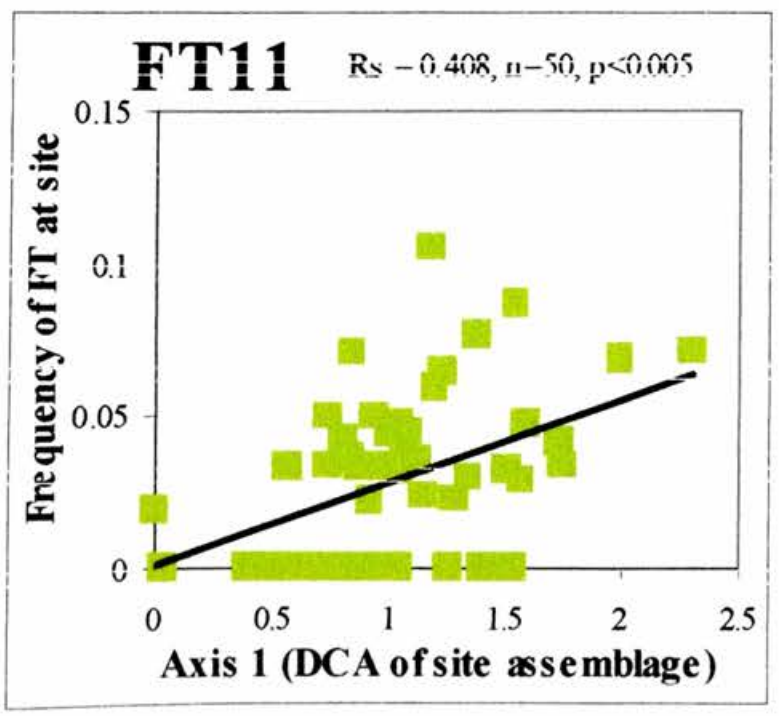


Figure 3.5: *Axis 1 vs. Axis 2 of a Detrended Correspondence Analysis (detrended by segments) carried out on trait category data set for 100 species found on derelict land site exhibiting distribution of different functional types. Colour coding denotes correlation with successional gradient in Fig 3.4. Dark orange = Pioneer FTs; Purple =Intermediate FTs and Green = Later successional FTs.*

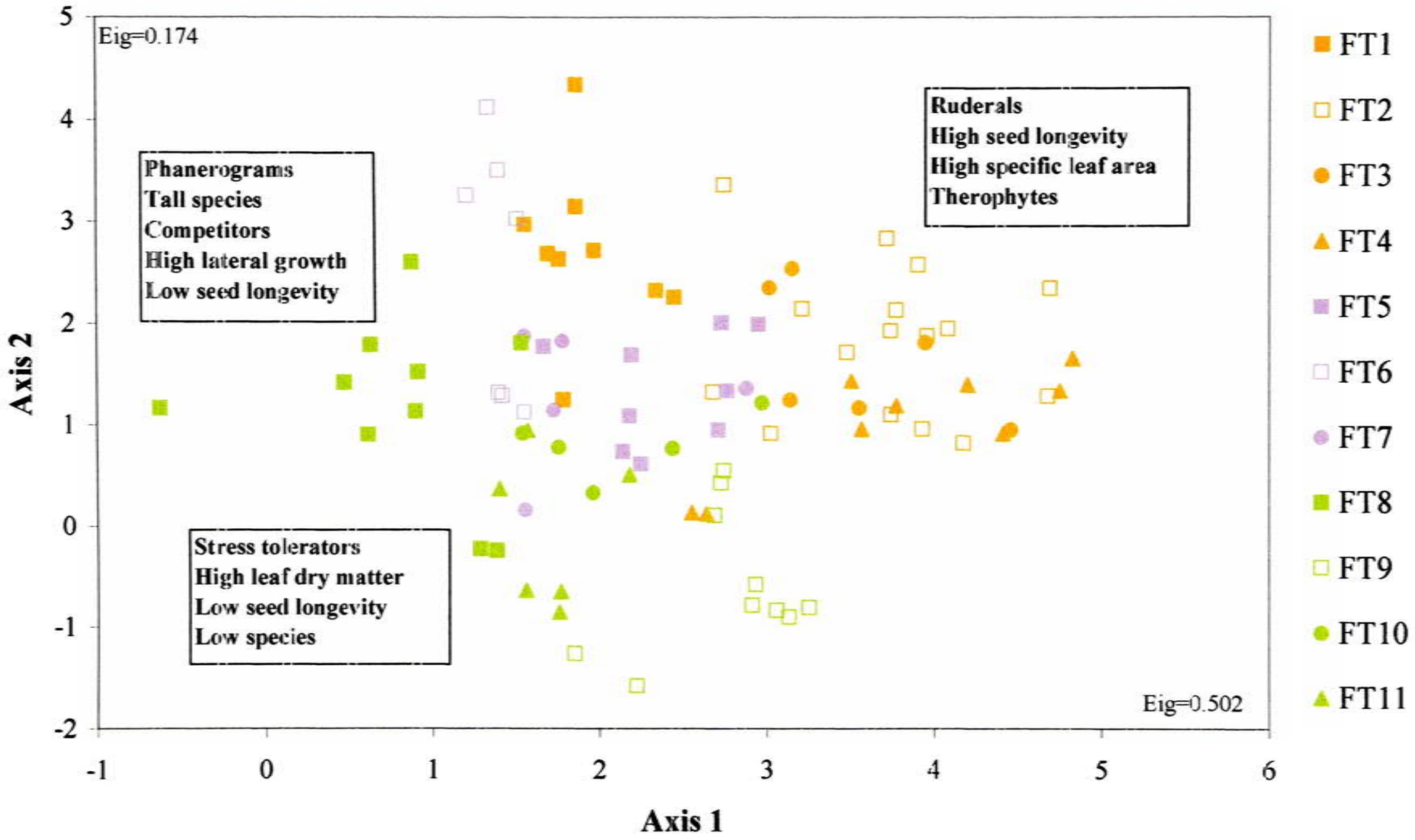


Figure 3.6(a): *Axis 1 vs. Axis 2 of a Detrended Correspondence Analysis (detrended by segments) performed on trait category data set for species found in pioneer FTs (FT1-4)*

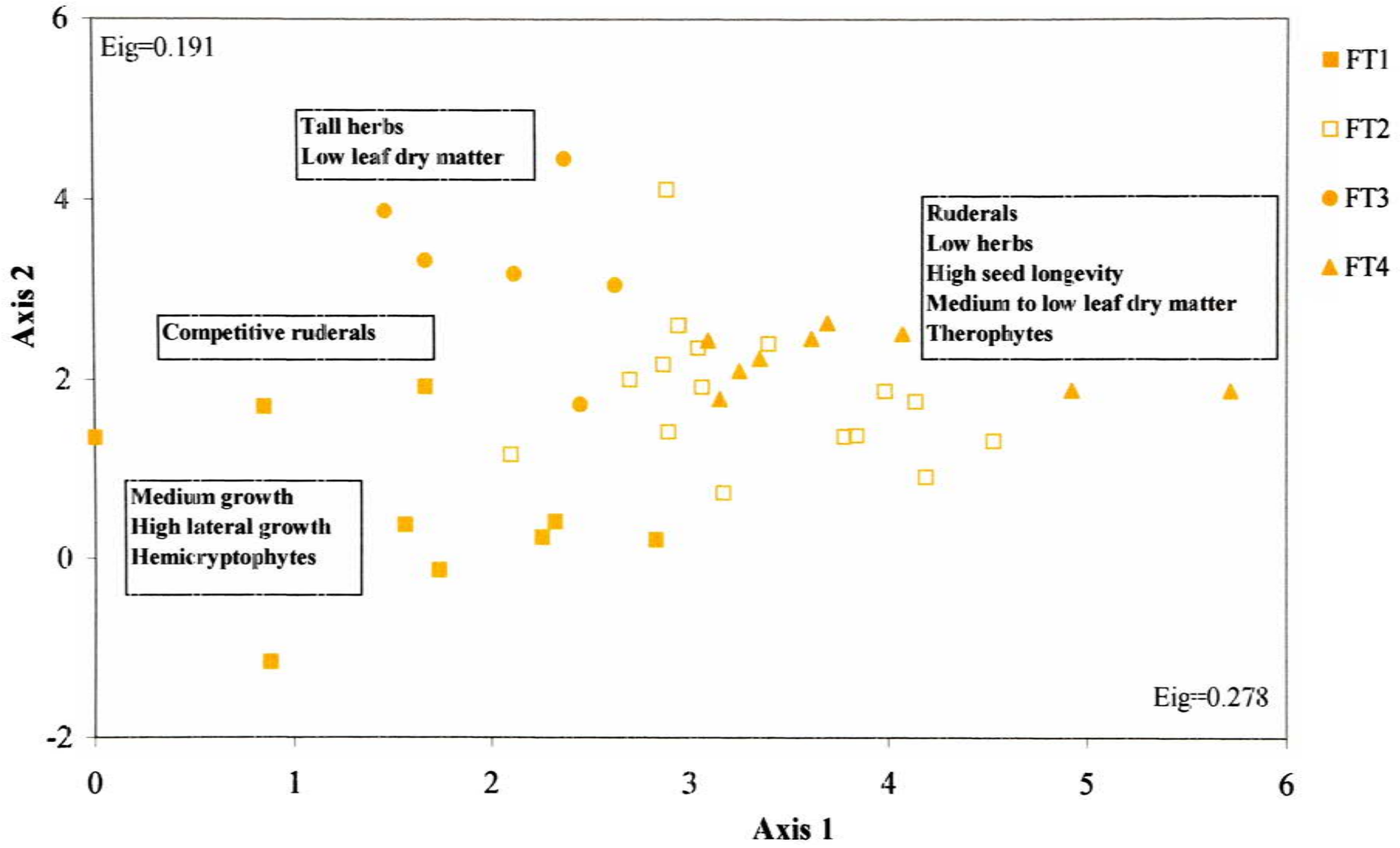


Figure 3.6(b): *Axis 1 vs. Axis 2 of a Detrended Correspondence Analysis (detrended by segments) performed on trait category data set for species found in intermediate FTs (FT5-7)*

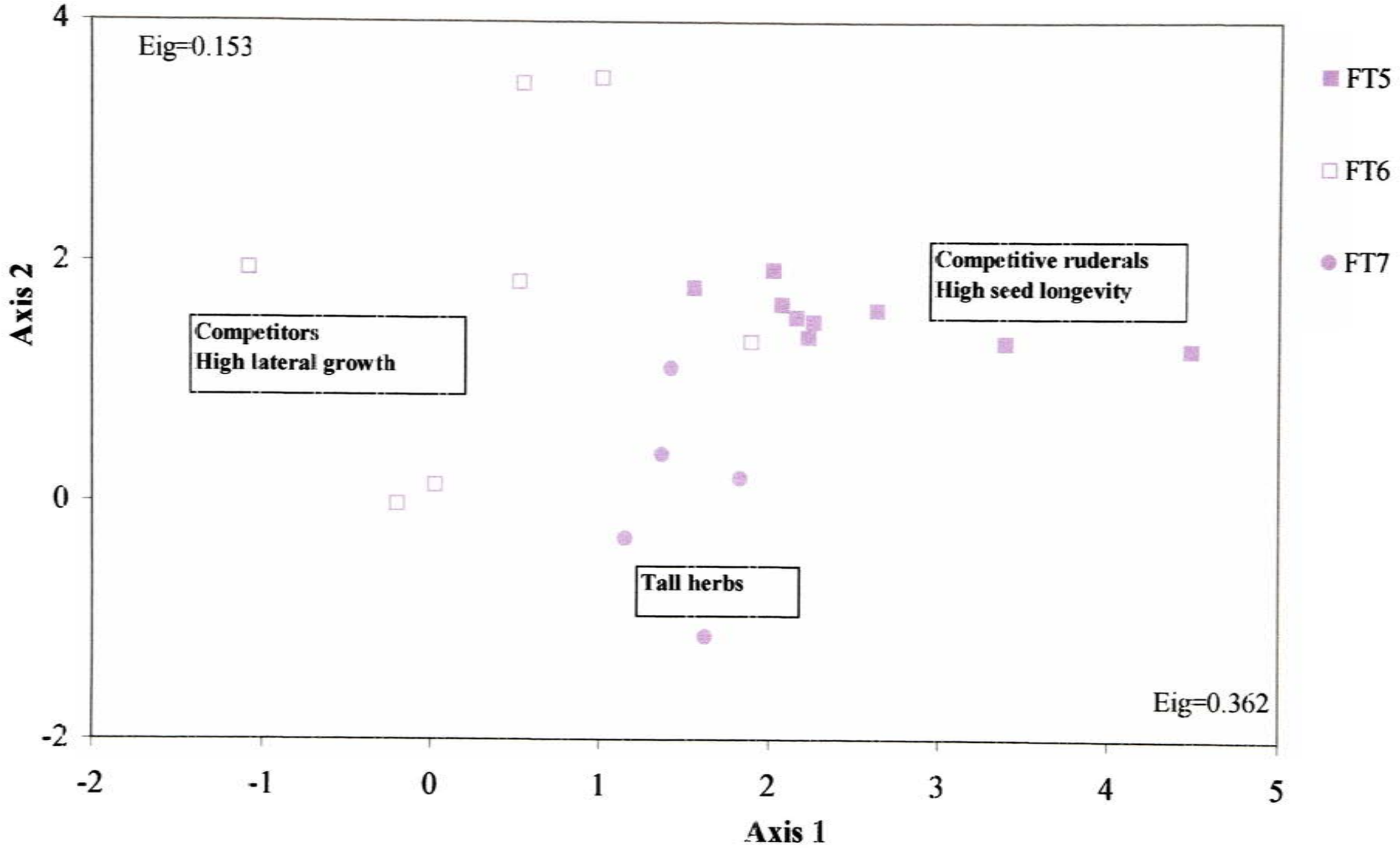
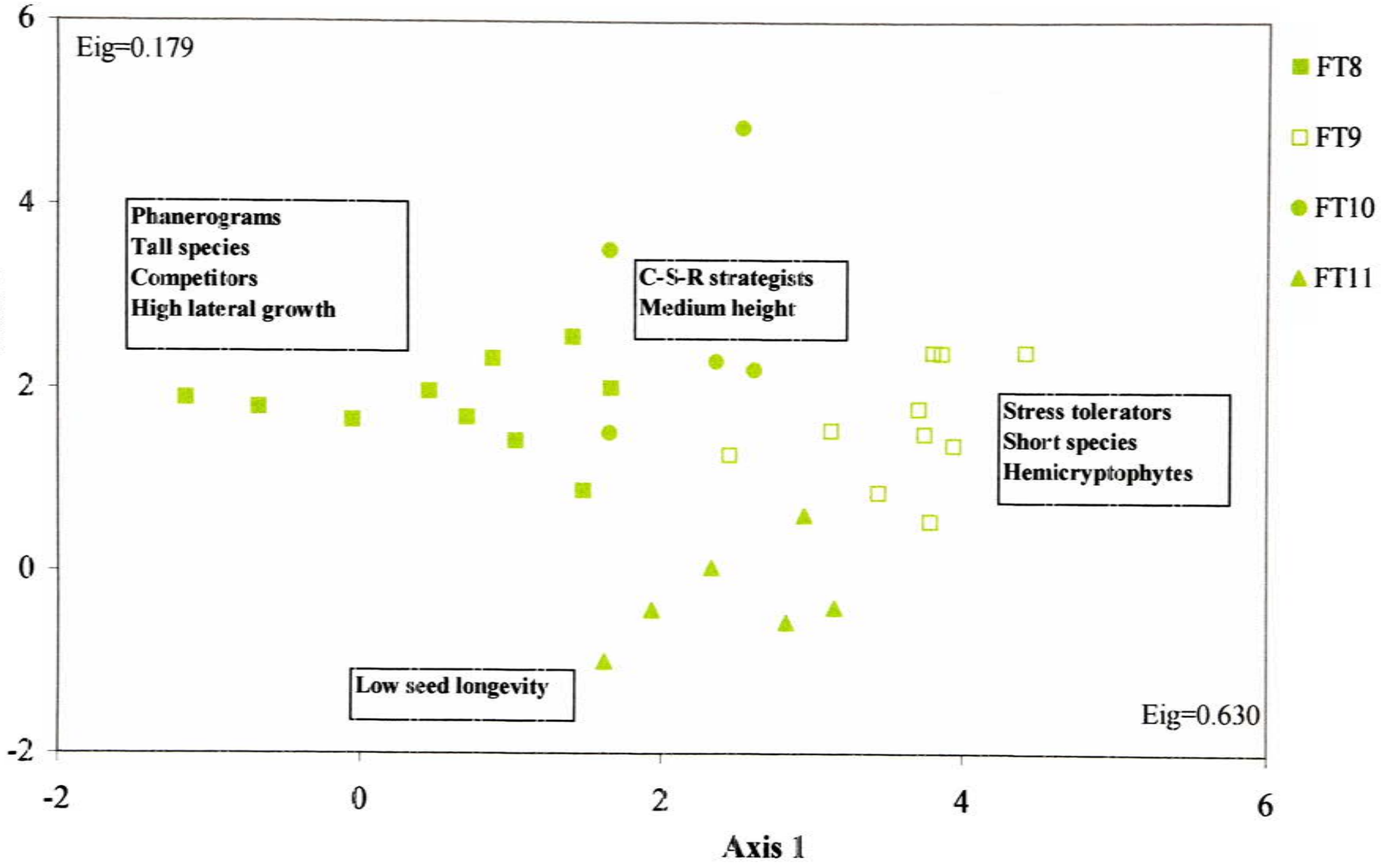


Figure 3.6(c): *Axis 1 vs. Axis 2 of a Detrended Correspondence Analysis (detrended by segments) performed on trait category data set for species found in later successional FTs (FT8-11)*



3.6 RESULTS (3): ENVIRONMENTAL FACTORS AND SUCCESSIONAL PATHWAYS

To investigate changes in the environment associated with the successional gradient the British Ellenberg values (Hill *et al* 2000.) were used as indirect measures of light, moisture, pH and nitrogen. The mean Ellenberg values for the species in each quadrat of the site were calculated, in order to reflect the environmental heterogeneity within sites. The mean value of all species found at the site was also calculated to give an estimate of the prevailing environmental conditions found there. The Ellenberg measures for quadrats and sites were then tested against the respective first axis scores for quadrat and site DCAs using Spearman's Rank correlation (Table 3.5).

Results from quadrat data and site data both indicated a significant decline in soil fertility along the successional gradient ($p<0.001$ Quadrat data, $p<0.001$ Site data). Both measures also pointed to a slight increase in acidity along the gradient, although this was only significantly shown by the larger quadrat data set ($p<0.001$ Quadrat data). Quadrat results also showed that species typical of moister soils were found at greater frequency at sites at an earlier successional phase ($p<0.001$ Quadrat data). The availability of light was not shown to change consistently with the successional gradient in either direction.

The mean Ellenberg values were calculated for species in the pioneer (FT1-4), intermediate (FT5-7) and later successional (FT7-11) categories. These means were then compared by ANOVA and Tukey test multiple comparison tests (Table 3.6). Early successional and intermediate groups showed broadly similar values for all variables but were significantly different to the later successional group for Nitrogen and Reaction, reflecting the finding of Table 3.5. Deviation from the group mean was highest in the FTs in the later successional group for all variables except moisture (intermediate) (also shown in Table 3.6 above). These findings suggest either that there is greater environmental heterogeneity at older sites, or that other factors are more important in determining the dominant species present at younger sites. Significant comparisons between mean values for functional types in the same successional category are shown in Table 3.7 and Figure 3.7. Differences between later successional types were most significantly illustrated for light and moisture. However, FT10 and FT8 for nitrogen ($p=0.08$) and FT11 and FT8 for pH ($p=0.11$) also showed quite distinct values. In

Table 3.5: *Spearman's Rank correlations between mean Ellenberg value and DCA axis 1 score for (a) quadrat and (b) site data (**** p<0.001)*

(A) QUADRAT DATA (N=1056)

Ellenberg variable	Correlation between mean quadrat Ellenberg score and quadrat DCA axis 1 score (Axis1 Eig. 0.430)	
	Rs	p-value
LIGHT	-0.006	n.s.
MOISTURE	-0.190	****
REACTION	-0.251	****
NITROGEN	-0.630	****

(B) SITE DATA (N=50)

Ellenberg variable	Correlation between mean site Ellenberg score and site DCA axis 1 score (Axis1 Eig. 0.212)	
	Rs	p-value
LIGHT	-0.131	n.s
MOISTURE	0.038	n.s.
REACTION	-0.241	n.s.
NITROGEN	-0.691****	

Table 3.6: *Comparison of mean Ellenberg values for species in pioneer, intermediate and later successional functional types using One-way Analysis of Variance and Tukey Multiple comparison tests († p<0.1, **** p<0.001)*

FUNCTIONAL TYPE GROUPINGS	MEAN ELLENBERG VALUE OF COMPONENT SPECIES			
	LIGHT	MOISTURE	REACTION	NITROGEN
Pioneer FT1-4	7.02 +/- 0.65	5.02 +/- 0.80	6.57 +/- 0.78	6.07 +/- 1.24
Intermediate FT5-7	6.90 +/- 0.55	5.30 +/- 1.41	6.70 +/- 0.86	6.10 +/- 1.25
Later successional FT8-11	6.55 +/- 1.27	5.11 +/- 0.78	5.72 +/- 1.13	4.50 +/- 1.44
ANOVA	†	n.s.	****	****
Pioneer vs. Intermediate	†	n.s.	n.s.	n.s.
Pioneer vs. Later successional	n.s.	n.s	****	****
Intermediate vs. Later successional	n.s.	n.s.	****	****

Table 3.7: *One-way analysis of variance and Tukey multiple comparison tests (p<0.05 level) comparing mean Ellenberg values for Functional Types associated within the same succession phase († p<0.1, * p<0.05, ** p<0.01, *** p<0.005).*

	COMPARISON BETWEEN PIONEER FTs (FT1-4)		COMPARISON BETWEEN INTERMEDIATE FTs (FT5-7)		COMPARISON BETWEEN LATER SUCCESSIONAL FTs (FT8-11)	
	ANOVA	TUKEY	ANOVA	TUKEY	ANOVA	TUKEY
LIGHT	n.s.	-	n.s.	-	*	FT8 > FT10
MOISTURE	n.s.	-	n.s.	-	***	FT10 > FT8, FT9 & FT11
REACTION	*	FT1 & FT3 > FT2 & FT4	n.s.	-	n.s.	-
NITROGEN	n.s.	-	n.s.	-	†	-

Figure 3.7: *Mean species light, moisture, reaction and nitrogen Ellenberg values within each functional type.*

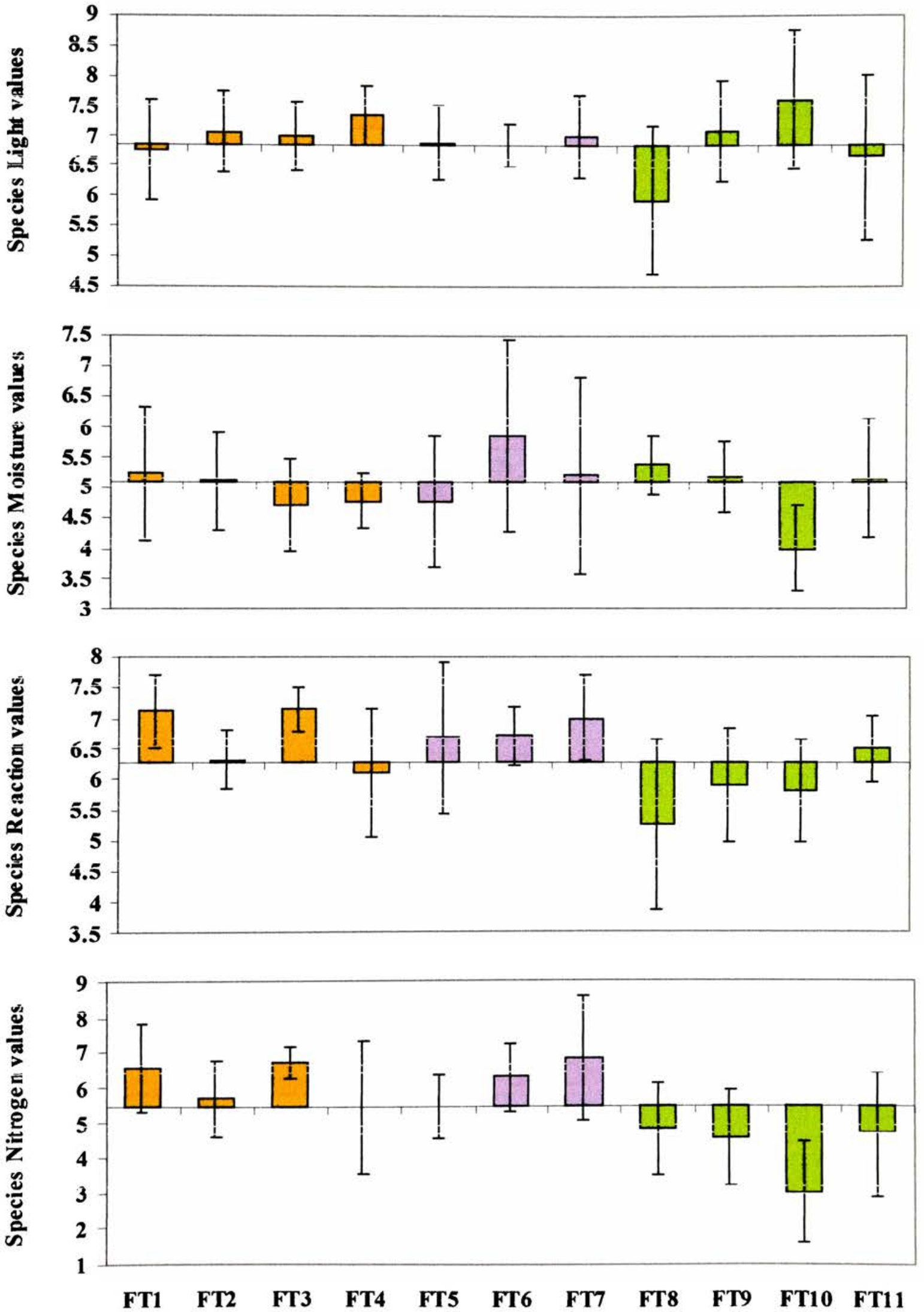


Table 3.8: *Schematic diagram illustrating the range of ecological tolerances exhibited by the component species of each functional type.*

FTs	AGE				LIGHT				MOISTURE			REACTION			NITROGEN				
	Young sites	Intermediate age	Older sites	Deep shade	Shaded	Light shade only	Well lit	Full sun	Dry	Slightly Dry	Moist	Damp	Acidic	Moderately acidic	Weakly acidic	Weakly basic	Infertile	Low fertility	Intermediate	Fertile	Highly Fertile
FT1	✓	✓				✓	✓	✓		✓	✓				✓	✓			✓	✓	✓
FT2	✓					✓	✓	✓		✓	✓			✓	✓			✓	✓	✓	
FT3	✓					✓	✓			✓	✓				✓	✓			✓	✓	
FT4	✓	✓				✓	✓			✓	✓		✓	✓	✓			✓	✓	✓	✓
FT5	✓	✓	✓			✓	✓			✓	✓	✓		✓	✓	✓		✓	✓		
FT6	✓	✓	✓			✓	✓			✓	✓	✓			✓	✓			✓	✓	
FT7	✓	✓	✓			✓	✓			✓	✓	✓		✓	✓				✓	✓	✓
FT8		✓	✓	✓	✓	✓				✓	✓		✓	✓	✓			✓	✓		
FT9			✓			✓	✓	✓		✓	✓			✓	✓			✓	✓		
FT10		✓	✓				✓	✓	✓					✓	✓		✓	✓			
FT11			✓			✓	✓			✓	✓				✓	✓		✓	✓	✓	

contrast the seven pioneer and intermediate FTs exhibited closely overlapping environmental ranges. The only significant, or close to significant, difference within or between these groups occurred for pH.

A condensed description of environmental tolerances based on Ellenberg mean and standard deviation within each functional group is also given in Table 3.8. Later successional types show coverage across 16 of the 18 qualitative bands across which all functional types stretched. In contrast, both pioneer (13) and intermediate (12) types and indeed the combination of both (14) showed less differentiation.

3.7 DISCUSSION

The ordinations carried out on both the abundance and site assemblage data in this study emphasised that the successional gradient could explain the majority of the variation between plants found on derelict sites. However the expected changes in plant traits were found to be illustrated more clearly along the first axis of the ordination output than they were against site age. This illustrates the problem of using age alone as an indicator of successional status on derelict sites. Small-scale haphazard disturbances are common on many sites post dereliction and may setback succession. Small fires, tipping and trampling can all create patches of vegetation at less advanced states of succession within older vegetation. The frequency and intensity of these disturbance events varies considerably between sites, particularly depending on how accessible they are. Substrate can also play an important role in how fast progression through successional stages takes place (Chapter 2, Sukopp *et al.* 1981). Over heavy clay for instance, the vegetation found after 15 years may be similar to that found over a loamy urban topsoil after only 6-10 years after dereliction (Chapter 2). Consequently, rather than displaying synchronicity, sites representing similar time spans of regeneration display a mosaic of successional stages (Prach *et al.* 1999).

It was notable that some patterns of change in plant traits observed in this study were not as clear cut as those observed in other temperate studies. In some instances the results verged towards being mutually conflicting. The strong relationship between FT8 (principally trees and shrubs) and age was to be expected and yet correlated traits like heavy seeds and animal dispersal showed no significant increase across the whole data set. Some of these conflicting

results may be explained by the manner in which regeneration is disrupted and altered in the urban landscape. Within a natural or semi-natural landscape bird numbers are particularly high within woodland and especially scrub and older grassland areas. Many animal species may also be found in these habitats. Importantly for regeneration there is also a continuity of habitat enabling animal species to colonise or utilise regenerated habitats after some vegetation development. Under these conditions, the transfer of climax species seeds to the regenerating patch can occur reasonably freely. In contrast, later successional habitats in cities are often highly fragmented and the movement of animals and birds may between them may perhaps be reduced. Regeneration on derelict patches in cities may therefore occur under circumstances where the arrival of seeds of natural climax community species is either absent or limited. This is emphasised by the strange communities observed on many urban sites of 50 or more years of age. Gilbert (1989) noted that in Birmingham these sites commonly hold a mixture of *Fraxinus excelsior, Crataegus* spp., *Salix* spp., *Viburnum opulus, Laburnum anagyroides, Ligustrum* spp. and *Malus* spp. There is shortfall in the number of typical climax species that might occur in more natural locations or indeed within a rural setting. This is perhaps particularly so for some of the heavier-seeded species such as *Quercus* spp. and *Fagus sylvatica,* which can only be readily dispersed over distance by the larger woodland bird species such as jays. Early successional sites in cities also indicate different patterns of recruitment. Woody species with edible fruits often occur quite early in urban successions. *Cotoneaster* spp., *Cornus* spp. and *Sorbus aucuparia* were among those recorded frequently on recently derelict sites in Birmingham. In some instances, these shrubs and trees may originate from gardens previously on the sites, but they were also often present on sites where the previous land use does not suggest plantings would have occurred. The importance of dispersal by birds from gardens or urban parks to new site seems therefore to be high.

The most typically urban bird species, including pigeons and starlings are often less arboreal in habitat and frequent open locations more often. These species seem relatively uninhibited in their movement across all but the most densely concreted areas of a city. Furthermore, remnants of past human constructions or occupation on derelict sites such as fence posts or old garden trees left *in situ* make suitable perches for other birds on newly derelict sites. In natural circumstances, many woodland birds will not commonly venture onto successional seres until the vegetation structure has developed sufficiently to provide these viewing points.

The gradient of potential for bird dispersal from early to mid succession may therefore be less extreme in cities.

Anthropogenic dispersal is also likely to be important. Casually discarded items of fruit and, on a larger scale, tipping of garden waste on open access sites, means the recruitment potential for woody species on newly derelict sites can be higher than in natural conditions.

Regeneration potential in cities at the outset is therefore determined by a combination of the available seeds in the soil seed bank after dereliction, the proximity of seed sources such as gardens and remnant patches and anthropogenic dispersal. While the initial seed bank, if there is one, is likely to be dominated by small seeds (see Chapter 4), anthropogenic dispersal selects for a variety of seed sizes (Hodkinson & Thompson 1997) and proximate seed sources may be diverse in seed size. The natural succession from predominantly small-seeded species to large-seeded species is perhaps therefore not as clear-cut in cities.

In this study, environmental factors explained differentiation between the functional attributes of species much better for vegetation at older sites. At this point in a succession, biotic interactions are much more intense and the outcome of interspecific competition in particular tends to shape the communities found. In these highly competitive circumstances, the environmental niche of many later successional species may be a very narrow range in which they are at greatest functional advantage over their rivals. In contrast, the different strategies employed by successful pioneer species have evolved in a way that maximises their ability to overcome the spatial and temporal barriers to successfully colonising newly denuded patches. The significance attributed to surrounding vegetation and size of disturbance for determining the pathway of a succession (van Andel *et al.* 1993; Bazzaz 1997) emphasises the importance of speed of arrival to successful colonisation. Persistent seed banks, high reproductive capacity, short life-span, light seeds and plumed seed attachments are all characteristics which can benefit a pioneer but which are not directly related to light, moisture, nutrients or pH in this instance.

The smaller impact of environmental factors on the composition of pioneer vegetation stands is also likely to be centred on the ability of colonising species to exploit a broader niche than species of mature vegetation (Parrish & Bazzaz 1982). Investigation of seed banks on derelict sites in this region support this theory. The soil under these sites has been shown to be

dominated by a very small pool of species, which occur widely irrespective of environmental factors or the type of dereliction (Chapter 4). The implication of these results is that a greater diversity of functional types become established as succession progresses. At first consideration this finding may be thought to indicate a divergence of vegetation during succession, in sharp conflict with the convergent outcomes described by other authors working on highly artificial substrates (e.g. Hatton & West 1987, Borgegard 1990) and indeed within the community analysis of this study (Chapter 2). However, if viewed from the species angle, it is clear that this is not the case. The number of component species within the pioneer types is very high relative to the later successional groupings. The high number of co-existing early successional species with different combinations of colonisation-adapted functional attributes and overlapping environmental tolerances found is probably a consequence of two factors. Firstly, there is considerable variation across the urban zone in the proximate recruitment pool of species, due perhaps primarily to landscape heterogeneity but also due to the way in which the type of dereliction influences which seeds are already present on the flattened site. Secondly, it is likely that actual establishment represents a chance selection from this available pool of species which provides a degree of lottery recruitment (*sensu* Sale 1977) that may delay the onset of competitive exclusion (Connell 1978; Huston 1979). In contrast on older sites, the vegetation is largely dominated by coarse mesotrophic grassland and by scrub. Diversification of functional attributes does exist, however, at the site level because of the presence of occasional patches of shallow soil or dry rubble, which has enabled the persistence of stress tolerators. In some instances where the substrate has proven particularly unfavourable to the development of extensive competitive stands, these stress-tolerating types may be dominant. However, it should be noted that if succession was allowed to progress on most of these sites, then much of the smaller scale edaphic heterogeneity would decline. Trees, because of their wide canopies and large amount of leaf litter, would gradually lead to a harmonisation of soil types and the elimination of stress tolerators. This process was partly demonstrated by the community analysis of the previous chapter that demonstrated that on the harsher rubble sites, the coarse grassland type vegetation was prevented from becoming dominant until after trees had been present on the site for some time. It is apparent therefore that the appearance of functional divergence exhibited by this study is in part a feature of the middle stages of the succession and not the later stages.

The large amount of research carried out into successional processes over a wide range of human-disturbed habitats in Central Europe (e.g. Prach 1987; Szegi *et al.* 1988; Prach *et al.* 1993; Prach & Pyšek 1994a) has enabled the development of an expert system (SUCCESS) to give predictions on seral stages and dominant species based on simple information regarding geography, relief, substratum, moisture, nitrogen and surrounding habitats (Prach *et al.* 1999). Within the UK a similar system (SETSARIO) has been developed for predicting re-growth on set-aside land (Hill 1990), while another (TRISTAR) can predict the impact of management on secondary systems (Hunt *et al.* 1991). Such an expert system, capable of predicting successional pathways on different urban derelict sites in Britain, could offer considerable potential benefits to landscape planners and conservationists. Ultimately however development of such software can only be comprehensive and reliable if the information it receives is both detailed and accurate. Further research on a broad range of derelict successions with good geographical coverage would be necessary to make such a system feasible here.

3.8 CONCLUSIONS

a) The use of site age alone to estimate the successional state of derelict sites in the West Midlands is unreliable due to differences in the rate of regeneration probably determined by substrate type and localised disturbance events.

b) The absence of clear changes in the abundance of both animal-dispersed and heavy-seeded species during succession on derelict urban sites may be indicative of modifications to the available recruitment pool available within urban areas.

c) The extent of differentiation between the ecological amplitudes of functional types varies between successional stages in a manner consistent with existing theories of succession and niche exploitation.

CHAPTER FOUR

SEED BANKS ON URBAN DERELICT SITES IN THE WEST MIDLANDS, UK

SUMMARY

Existing knowledge about the ecology of seed banks was tested on a previously neglected habitat. Twenty-nine derelict land sites within the West Midlands conurbation were surveyed and their seed banks investigated using greenhouse emergence methods. The findings of the wider body of seed bank research were used to establish hypotheses regarding 1) depth distribution in the soil and 2) changes along successional gradients. The importance of previous land use on derelict sites was also examined. Results suggest that seed banks in these habitats largely conform to expected patterns. Seed density, species diversity and similarity with the vegetation all decline significantly with increasing soil depth. Seed density and similarity also decline along a successional series. Changes in the relative representation of certain plant traits: seed longevity, seed mass and shape and life history also show expected changes along both depth and successional gradients. In general, the seed banks at these sites were found to be dominated by abundant numbers of a few well-distributed species, irrespective of previous land use. The type, frequency and intensity of disturbances typical on these sites are also considered in respect of their impact on seed bank ecology.

KEYWORDS: *Derelict land, Seed longevity, Succession, Urban ecology.*

4.1 INTRODUCTION

Derelict land in urban areas plays a significant role in contributing to diversity at a local level (Teagle 1978; Greenwood & Gemmell 1979; Shepherd 1994). The species assemblages and communities present show considerable diversity between sites. Furthermore, each city and zones within them often have their own distinctive component species (Pyšek 1998). This diversity can be closely associated with differing historical land use that determines the nature of the considerable anthropogenic modifications to the urban environment (Sukopp & Werner 1983).

Attempts have been made to describe and classify some of the floristic associations found on these artificial habitats, most notably in central and eastern Europe (Kornas 1978; Janecki & Sawczuk 1984; Kubikova 1990), and to a lesser extent within the UK (Shepherd 1994; Rodwell 2000). However a complete description of any plant community will require the inclusion of the buried viable seeds in the soil (Major & Pyott 1966). At the current time research on seed banks within an urban context has been very limited. Consequently, a detailed study of the seed banks found under urban derelict land will contribute significantly to current understanding of both urban ecology and seed banks. The framework of this study is to test several predictions made from existing knowledge of seed bank ecology.

Changes in seed bank composition associated with successional gradients

The vegetation found growing on derelict land in the study region strongly reflects the time since abandonment or demolition. As a collection of sites, they represent a successional series from early pioneer to mid-successional status. Previous work on successional series (Donelan & Thompson 1980) has emphasised that, by definition, there will be a gradient of increasing stability towards the climax community.

Seed bank research on stable habitats, which include for example those on mature woodlands (Ooosting & Humphreys 1940; Olmsted & Curtis 1947; Livingston & Allessio 1968; Strickler & Edgerton 1976) and those on old semi-natural grasslands (Chippindale & Milton 1934; Champness & Morris 1948; Major & Pyott 1966), consistently find that seed density is low. Several studies (Oosting & Humphreys 1940; Thompson & Grime 1979; Brown & Oosterhuis

1981) have also demonstrated and that there is poor correspondence with the associated vegetation. The poor correspondence between the seed bank and the above ground vegetation has prompted numerous authors to observe that seed banks do not provide much restoration potential for stable habitats (e.g. Hutchings, Graham & Booth 1990; Milberg 1992; Bakker *et al.* 1991; Akinola, Thompson & Buckland 1998).

A number of studies which have actually examined the progression of seed banks along successional gradients have also shown a recurring pattern, where both seed density and similarity with the flora decline from an early successional peak (Oosting & Humphreys 1940; Livingston & Allessio 1968; Roberts & Vankat 1991; Donelan & Thompson 1980; Nakagoshi 1984). Under rarer circumstances, where the early stage of succession is maintained more by stress than disturbance, there will however be a smaller seed bank (Donelan & Thompson 1980). Seed bank density is reduced over the period of a normal secondary succession by a combination of two factors a) direct reduction of the seed bank in the soil due to loss of viability, germination, predation and fungal attack (Roberts 1970); and b) reduced deposition associated with a change in vegetation strategy along the successional gradient (Donelan & Thompson 1980). However, in some instances large persistent seed banks may remain in the soil of climax or plagioclimax ecosystems some time after the progression of secondary successions. Conifer plantations that have been planted over old heathlands often contain high densities of *Calluna vulgaris* seed within their soil (Hill & Stevens 1981). Similarly, grasslands may have high numbers of arable weed seeds in the soil if cultivation has occurred in the past (Schenkeveld & Verkaar 1984). In this situation seed numbers of weed species can remain high for many years, particularly if they are buried deeply (Kropac, Havranek & Dobry 1986; Conn & Farris 1987).

In general, the flora of frequently disturbed and younger habitats will be a closer representation of the seed bank because weedy or ruderal species, which are well represented in the soil, are the first to take advantage of the fresh gaps that appear. The economic interest in arable weeds has provided the impetus for considerable work on seed banks on this unstable system and similarity between the seed bank and the weeds in the fields above has been found to be high (Jensen 1969; Wilson, Kerr & Nelson 1985). Much larger seed banks are also associated with cultivated systems. Seed banks under arable land record some of the highest densities in the literature (Brenchley & Warington 1930, 1933; Jensen 1969).

Heathlands, which frequently have sporadic fires in dry summers, also have large persistent seed banks of the dominant species in the vegetation, predominantly *Calluna* (Chippendale & Milton 1934; Miller & Cummins 1987). In the tropics, higher seed densities are also associated with regularly disturbed land (Ewel *et al.*1981; Young 1985; Young, Ewel & Brown 1987) and with secondary re-growth (Guevera & Gomez-Pompa 1972).

Therefore, it can be predicted that derelict urban sites will have relatively high seed densities, due to i) the young age of many sites and ii) the high frequency of disturbances at many sites due to fires, tipping and other human activities. These factors are also likely to increase the similarity between the seed bank and the vegetation. However, it is likely that lower seed numbers and declining similarity with the vegetation may be found at older sites and at those with more restricted access (e.g. boarded up sites) which help to minimise disturbance.

Seed characteristics in relation to longevity in the soil

Changes in the representation of different plant characters and life histories along environmental or temporal gradients are to be expected. These changes are linked directly to the manner in which selective forces change along the gradient in favour of different types of physiology or life history. Studies on the changes of life history attributes and physical characteristics along successional series have a long history (e.g. Clements 1916). Seed morphology is also strongly correlated with changes associated with succession. The link between increasing adult longevity and larger seed size as a succession progresses was established by Salisbury (1942, 1974). Large seeds are selectively advantageous in closed communities associated with late succession, most notably where light resources are low (Tilman 1988). In contrast, colonising species have notably small seeds and /or morphological adaptations to aid dispersal (Fenner 1987). To be most ready to take advantage at the onset of a secondary succession, the best strategy is to have a supply of dormant seeds already in the soil. Thompson & Grime (1979) concluded that it is competitor avoidance that has driven selection for seed dormancy mechanisms. Consequently, it is to be expected that pioneers on very new sites, which are quickly suppressed by later arriving competitors, to have seeds with higher longevity. Indeed, species present in transient habitats and those subject to spatially unpredictable disturbances, both features characteristic of urban derelict sites, will tend to have rapid and high reproductive potential, and/or the capacity for persistence as dormant

seeds in the ground (Hodgson & Grime 1990). Numerous 'bet hedging' models have also been produced which support increased success of persistence in variable less stable habitats (Venable & Brown 1988; Rees 1993; Cohen 1966; Bulmer 1984). In contrast, species typical of later successional states would not be expected to form long-term seed banks. This is borne out by the fact that most tree species produce transient or short-term persistent seed banks only (Howard & Ashton 1967; Marquis 1975; Conn, Cochrane & Delap 1984). The last twenty years have seen a considerable development in the understanding of the ecology of seeds in the soil. Most significantly, we now have a much better idea how the seed characteristics relate to the likelihood of incorporation into the long-term seed bank. Work on a range of habitats in Northern England enabled Thompson & Grime (1979) to determine four distinct types of seed bank strategy (I, II represented transient forms; III and IV representing both short to long term persistent). An additional contribution was provided by a key to the morphological traits and germination requirements of seeds (Grime & Hillier 1981) that predicted the seed bank group to which a species is likely to belong. Thompson, Band & Hodgson (1993) showed that of these traits shape and mass of seeds alone could allow predictions of persistent or transient status. Small spherical seeds were most favoured for persistence. More recently, by utilising a wide range of seed bank studies (see database in Thompson, Bakker & Bekker 1997), it has been possible to develop a continuous index of longevity based on known records of persistence (Bekker *et al.* 1998). The authors showed this to be significantly correlated with a combination of seed mass and variance in seed shape:

$$\textbf{Longevity} = \mathbf{M} \times \sqrt{\mathbf{V_s}}$$ (Where M = seed mass and V_s = variance in seed shape).

The seed longevity index produced provides a strong prediction for depth distribution of species in the soil (Bekker *et al.* 1998). This confirms the assumptions of many authors (e.g. Chippindale & Milton 1934; Moore & Wein 1977; Harper 1977, Kellman 1974; Brown & Oosterhuis 1981; Leck 1989) that seeds deeper in the soil are older and that short-lived seeds will not reach lower levels. Comparison of the seed banks of different depth cores have been carried out within numerous studies and consistently show a decline in seed density and species number (Numata *et al.* 1964; Kellman 1974; Moore & Wein 1977). Similarity with the flora has also been found to decline down the soil profile (Grandin & Rydin 1998). If the soil is relatively undisturbed, it appears that seeds of transient and short-term persistent

species quickly disappear down the soil depth gradient explaining these declines.

If seed banks on derelict sites conform to patterns observed in other habitats, it is expected that smaller and more rounded seeded-species are more likely to be found in large numbers in the soil and will form the great majority of the seeds found at greater depths. One of the aims of this study is to test changes in the representation of these key seed traits in relation to both successional gradients and depth profiles.

Spatial heterogeneity of seeds in the soils and sampling efficiency

The haphazard nature of disturbance events typical of these sites is likely to increase the spatial heterogeneity of seed diversity and density within the site. This can be viewed as additional to the natural patchiness in seed density distributions which occurs due to location of source plants and surrounding topography. Morphological characteristics, such as mucilaginous seed coats in *Juncus* sp. (Bigwood & Inouye 1988), or retention of seeds on inflorescences, may promote high density seed clusters. In addition, the interaction of local fauna may lead to aggregation e.g. under the feeding roosts of birds, or within ant hills (Kjellson 1985). Spatial patterns have important ecological ramifications, as clustering is likely to promote increased predation (Inouye 1980).

Very high levels of small-scale spatial variability have been recorded for seed banks from a number of studies (e.g. Schenkeveld & Verkaar 1984; Thompson 1986; Benoit *et al.* 1989). Consequently the requirements for an effective sample has been an important subject for seed bank researchers. A number of authors (Warr, Thompson & Kent 1993; Major & Pyott 1966) have pointed to a lack of samples in many previous seed bank studies. Actual published estimates for seed densities from studies of less than forty samples are not considered as reliable. Bigwood & Inouye (1988) found that lots of small samples were more precise than one or two large samples representing the same volume of soil, but that accuracy would still be closely determined by the total volume of soil sampled. Where a detailed investigation into within-site variation is not required, bulking samples can be an effective means of reducing greenhouse space and effort (Livingston & Allessio 1968; Brown & Oosterhuis 1981). Therefore an approach which utilises several bulked small cores to produce a sub-sample may be appropriate for studies, where an overview of the seed bank at the site-level is required (Gross 1990; Ter Heerdt *et al.* 1996).

For any major study, the requirements for a valid sample size almost certainly rule out the more labour intensive methods of seed bank analysis, notably flotation or hand-sorting (e.g. Malone 1967; Fay & Olsen 1978) which may be prone to under estimation of smaller species anyway (Ter Heerdt *et al.* 1996). In contrast, emergence methods are much simpler and more time-efficient for large-scale studies, although it is acknowledged that seeds may not emerge due to unfulfilled germination requirements that are known to vary greatly between species (Galinato & Van der Valk 1986). The benefits of the greenhouse emergence methods have also been greatly enhanced by the refinement of Ter Heerdt *et al.* (1996), which allows the examination of large volumes of soil in a much reduced greenhouse area.

The sampling methods laid out below provide for good estimates of mean total seed density across the derelict sites and for good estimates at each site of the densities of the most common species. Analysis of results for species number however is restricted within this paper to soil depth comparisons within a site, as it is accepted that between-site differences may be biased by differences in sampling size and sample area.

4.2 METHODS

Soil samples were collected from twenty-nine derelict sites within the West Midlands conurbation, UK. A brief description of the sites is given in Table 4.1. Samples were taken using a metal corer (5.5cm diameter). Cores were collected between 18-28 February 1999 and samples stored outside in black plastic bags for between 10-18 days before sieving. The timing of collection in late winter meant that those species that require cold stratification were likely to have received this.

At eleven sites, cores were taken to a depth of 10 cm and divided into two 5cm horizons. At the other eighteen sites, only the upper 5cm horizon was cored. Six sets of bulked cores were taken at each site. Each bulked set was collected from stratified sampling of the whole site. Total core sample number reflected site size on the following banded scale: up to 2000 square metres (90 cores), 2001 - 5000 sq metres (120 cores), 5001-10,000 sq metres (150 cores) and over 10,000 sq metres (180 cores). Bulked sets therefore consisted of 15, 20, 25, or 30 cores respectively depending on site.

Each of the six samples of bulked cores were wet-sieved separately through 4mm and

Table 4.1: *Location and description of study sites.*

ID	SITE	O.S. REF	DESCRIPTION	SUBSTRATE	AGE (YEARS)
1	Stourbridge	SO901851	Rough grassland at rear of new housing estate	Topsoil	12
2	Frankley	SO990784	Mound of clay near new development	Clay & hardcore	3
3	Platts Rd	SO895858	Small area of waste ground used for tipping	Brick rubble	5
4	Grange Rd	SP138812	Open patch of mostly bare ground at back of flats	Broken Tarmac	5
5	Solihull Arable	SP168818	Old field site	Agricultural Topsoil	4
6	Ackers	SP102844	Slope adjacent to ski slope	Refuse	6
7	Ruberry Shops	SO999782	Wide expanse of bare clay around new development	Clay & hardcore	4
8	Kings Heath	SP078796	Old parking area now derelict frequently used for fires	Broken Tarmac	5
9	Roman Rd	SK092004	Old field site	Agricultural Topsoil	4
10	Florence Rd	SP030875	Site of several demolished houses used for fly tipping	Brick rubble	8
11	Lake St	SK104007	Area of rough ground adjacent to road	Topsoil	5
12	Gravelly High	SP104916	Rough grassland adjacent to road	Brick rubble	15
13	Gravelly Low	SP099917	Area of rough grassland and scrub backing onto school	Brick rubble	15
14	Cradley Heath	SO949842	Old suburban garden and demolished house	Topsoil	17
15	Minworth	SP180920	Disturbed land beside sewage filter beds	Sewage	12
16	Small Bentley	SO990979	Recently formed embankment beside overflow brook	Refuse / topsoil	6
17	Large Bentley	SO988978	Common land used extensively for illegal tipping	Clay & hardcore	14
18	Hall Green	SP009944	Urban open space and common land	Refuse/Broken Tarmac	10
19	Sandy Lane	SP013951	Rough grassland used irregularly for grazing horses	Topsoil	20
20	Vincent Drive	SP042830	Site of demolished works, including metal works	Refuse	15
21	Saltwells	SO936878	Rough grassland adjacent to nature reserve	Topsoil	20
22	Cole Bank	SP099817	Rough grassland adjacent to road	Topsoil/BrokenTarmac	14
23	Samson Quarry	SO977891	Grassland over infilled quarry	Sandy	20
24	Turners Hill	SO969880	Infilled quarry site	Clay & hardcore	14
25	Burberry	SP099837	Old brickworks covered by clay cap	Clay & hardcore	8
26	Cole Round	SP141875	Old garages in housing estate	Topsoil	8
27	Mid Cole	SP132880	Rough grassland between R. Cole and backs of gardens	Topsoil	20
28	Bearwood	SP035867	Open area by demolished bridge along disused railway	Sandy	6
29	Wilson Rd	SP026872	Site of several demolished houses	Brick rubble	8

0.212mm mesh sieves to produce a final condensed volume for the germination trial. The condensed sample was mixed thoroughly and volume recorded and spread evenly over one sample seed tray (15 x 20 cm), containing a 3cm depth of sterile sand, to a maximum depth of 5mm. The volume of any excess condensed sample was recorded before being discarded. The 240 plastic trays were laid out in six randomised blocks. In addition, six control trays containing potting compost over sterile sand were laid out at intervals to test for contamination. Greenhouse conditions were maintained on a 16hr daylight regime with daytime temperatures controlled between 15-22°C. All samples were watered twice daily.

Chancellor (1966) and Muller (1978) were utilised to aid identification of herbs, shrubs and tree species. Hubbard's (1984) vegetative key proved helpful for grass species. Most seedlings were removed on identification from the seed trays, but, where density was high and early identification was not possible, seedlings were replanted in vermiculite/potting compost mix for growing on and later identification. Precautionary pesticide application was carried out twice during the survey period. The period allowed for germination for the main study was 14 weeks for each tray. At the end of this period, the soil layer in three trays from each site was broken up, thoroughly mixed, re-laid and left for a further month in order to investigate subsequent germination rates. For each of the twenty-nine sites, seed bank data were compared with vegetation data obtained from field surveys carried out on the sites in Chapters 2 and 3.

4.3 Results (1) Overview

At the end of the 14-week period 36,916 seedlings had germinated from the 240 trays. Herbivore impact was negligible. However a replicate for one site, Minworth, had dried out and was discounted from the analyses. It was possible to ascertain that at least 151 species were represented in the seed bank. Three of the more difficult, yet frequent taxa, were grouped because it was unfeasible to grow on every specimen. These were all species of *Sonchus* and *Epilobium* and all *Juncus* species with the exception of *J. bufonius*, which is distinguishable at quite an early stage. Consequently, 142 taxa were treated for analysis purposes. Where plant trait data are used in this study, the mean values of the species found from growing-on were calculated for each of these grouped taxa.

Plate 4.1: *Germinated seedlings from derelict land seed bank study growing on seed trays in greenhouse (May 1999).*

Plate 4.2: *Trays of individually potted seedlings grown on to confirm identification during seed bank study of derelict land (May 1999).*

Plate 4.3: *Germinated seedlings of (from left to right)* Artemisia absinthium, Artemisia vulgaris *and* Sisymbrium officinale.

Plate 4.4: *Germinated seedlings of (from left to right)* Trifolium pratense, Medicago lupulina *and* Trifolium repens.

Plate 4.5: *Germinated seedlings of (from left to right)* Anagallis arvensis, Stellaria media *and* Cerastium fontanum.

Plate 4.6: *Germinated seedlings of (from left to right)* Plantago major, Plantago lanceolata *and* Hypochaeris radicata.

Analysis of the extra seedlings that emerged in the four weeks after the main study suggested that the tail end of germination was being reached. Mean additional germination per site represented 6.75% (S.E. ± 0.67%) of total seedlings recorded in the main study. The maximum addition was 16.35% at Bearwood disused railway (Figure 4.1). Mean new species found at each site was 0.65 (S.E. ± 0.13) while the maximum number of new species found at any one site was 3 at Hall Green. Only one new species was found for the study as a whole. This was the large seeded legume *Lupinus x regalis*. Comparison of the commonest members of this family (Fabaceae) demonstrates that larger-seeded species showed significantly higher levels of additional germination than smaller-seeded species (Table 4.2). Three other commonly found species also showed considerably higher germination in the additional period. These were *Cardamine hirsuta* (22.7% of original total), *Hypericum perforatum* (58.2%) and *Veronica serpyllifolia* (26.0%).

Mean seed density varied considerably between sites (220 seeds m^{-2} to 46,000 seeds m^{-2}) (Figure 4.2) but at the highest levels was comparable to some of the highest densities recorded on arable sites (e.g. Brenchley & Warington 1930, 1933 (c.12,000-44,000 seeds m^{-2})). Considering that the sample depth for this measurement was only 0-5cm, it is feasible that at some sites, the seed density in the soil was in excess of 100,000 m^{-2} even accounting for declining numbers with depth. At most sites, seed density was considerably higher than those generally recorded at stable habitats in the UK, e.g. woodlands (Brown & Oosterhuis 1981 (610 m^{-2})) and grazed grasslands (Donelan & Thompson 1980 (1329 m^{-2})). Species number, which also fluctuated, showed no relationship with seed density (Figure 4.2).

Although 142 taxa were found across the 29 sites, closer examination revealed that the major proportion of each site's seed bank was composed of a small number of abundant species. Table 4.3 illustrates the most abundant species at each site, which in total represent at least 75% of total seedlings recorded there. This smaller pool of species, only 38 across all sites, demonstrate considerable overlap and suggest that recognition of different seed bank types in this study may be unrealistic.

Figure 4.3 shows the output of Detrended Correspondence Analysis (Hill 1979), for the species assemblages found within the sites in both the vegetation and the seed bank. The seed banks are clearly distinct from their associated above ground vegetation and are much more

closely grouped with each other. While the vegetation shows separation according to substrate type, the seed bank sites are tightly clustered together with little between-site variation. Indeed, where sites representing substrate types typical of the urban zone are grouped (Figure 4.4) the overlap becomes even more pronounced as the 10 most abundant taxa in the survey (Table 4.4) dominate the seed bank of each type. Each of these ten most abundant species is found in all the four common substrate types.

A comparison was made between the traits of component species of the seed bank study; the corresponding vegetation study; and the regional species pool (Table 4.5). The fifty most abundant species in each were compared. This was determined by using average density for the seed bank; number of regional records for the regional pool; and a combination of quadrat frequency and cover for the field study. For each subset, the representation of plant traits was determined from published and unpublished sources. Traits were analysed using a one-way analysis of variance and Tukey multiple comparison tests.

For all traits, there was a gradient in mean values from the derelict seed bank through to the regional pool, with the derelict flora somewhere in between. Most of the differences found between the subsets can be accounted for by habitat alone. Generally, higher values for specific leaf area (and ecologically related lower values for dry matter) were found for the common derelict species relative to the regional pool. Similarly, the most significant reduction in height occurs between the regional flora and the derelict subset. Significant differences between the derelict flora and the seed bank were restricted to seed longevity and the closely correlated life history. The seed bank was more heavily represented by annuals and long-lived seed species (see also Figures 4.5 & 4.6).

Table 4.2: *Additional germination in Fabaceae family (** % doubled as only half trays continued in extra 4 weeks)*

SPECIES	14 WEEK TOTAL SEEDLINGS	EXTRA 4 WEEK TOTAL SEEDLINGS	% ADDITION (ADJUSTED**)	LOG SEED WEIGHT (mg)
Trifolium dubium	159	8	10.1%	-0.64
Trifolium pratense	91	0	0.0%	0.14
Trifolium repens	400	13	6.5%	-0.25
Medicago lupulina	364	4	2.2%	0.10
Vicia hirsuta	89	19	42.4%	0.67
Vicia sativa	42	20	95.2%	1.08

Figure 4.1: *Additional seed density and species number recorded at each site in a further four week period following the three-month study. Only half the seed trays were used and additional germination has been doubled to account for smaller sample size.*

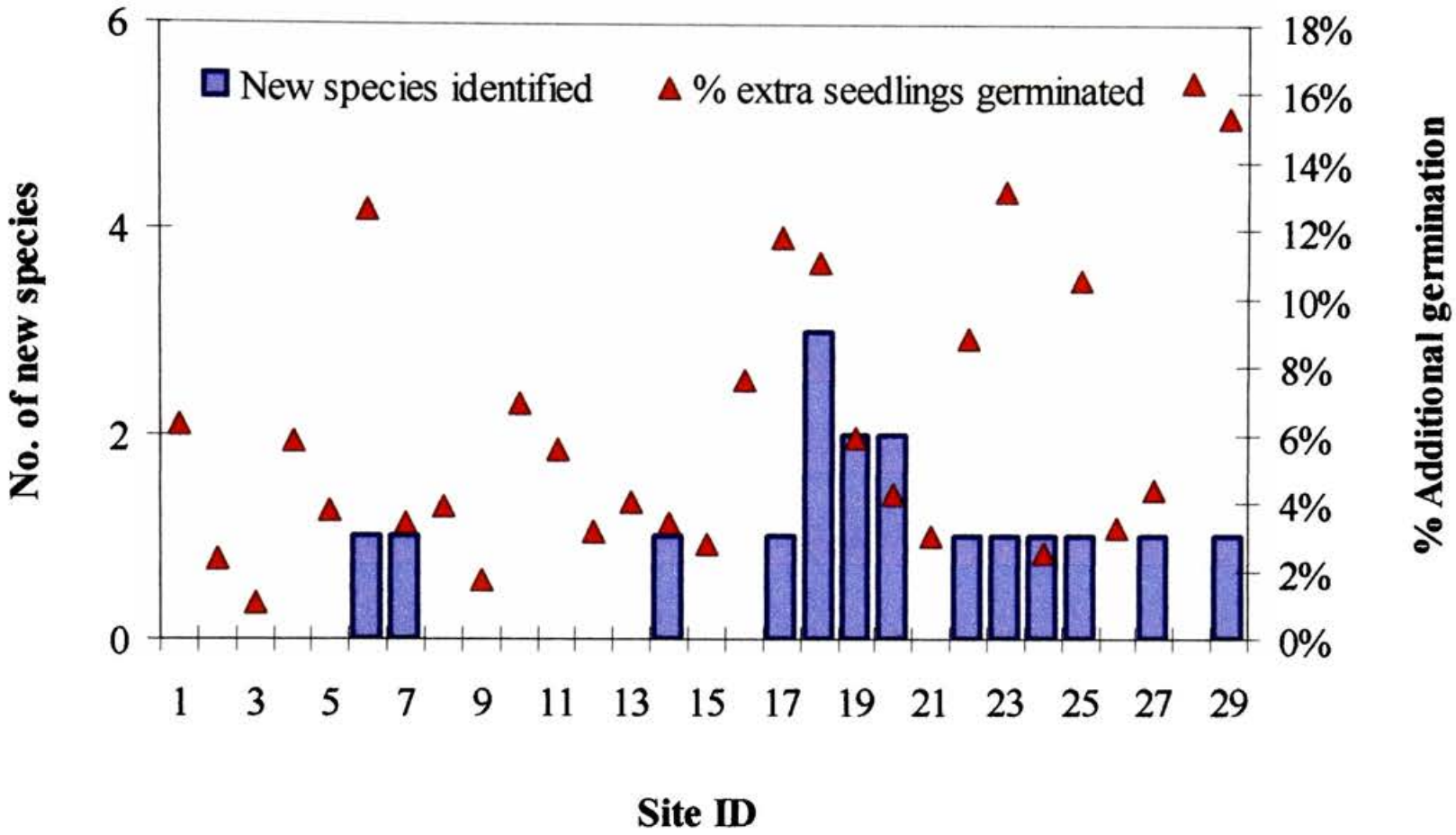


Figure 4.2: *Germinated seed numbers recorded during seed bank study of 29 derelict land sites in three month germination period March-May 1999. (Error bars represent standard deviation about six samples).*

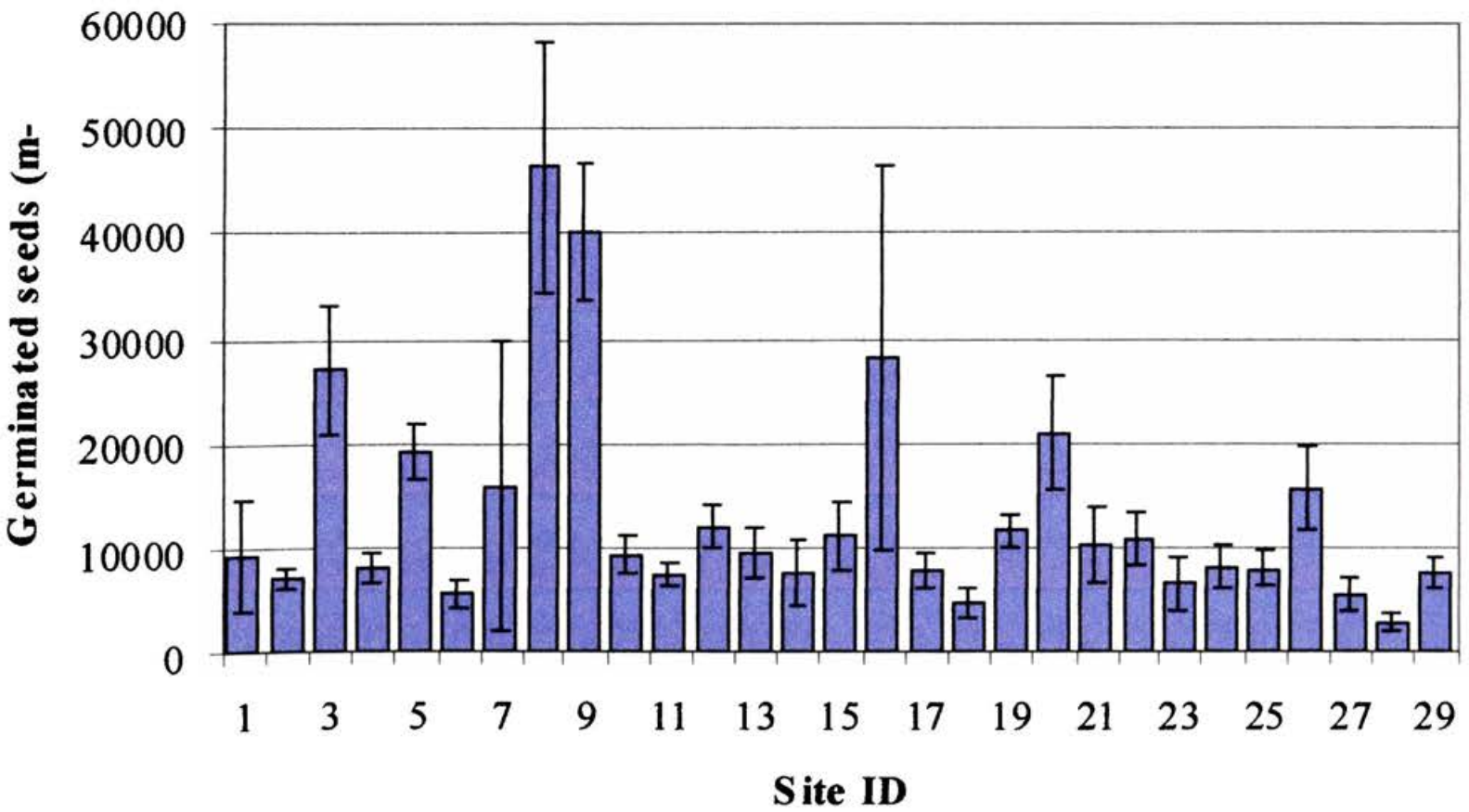


Table 4.3: *Dominant species in seed bank at study sites. Species marked by an asterisk are the most abundant found at a site and combined contributed to at least 75% of total seedlings germinated. (Full species list contained in Appendix VII)*

	1) Stourbridge	2) Frankley	3) Platts Rd	4) Grange Rd	5) Solihull	6) Ackers	7) Ruberry	8) Kings Heath	9) Roman Rd	10) Florence	11) Blake St	12) Gravelly H	13) Gravelly L	14) Cradley	15) Minworth
Achillea millefolium				*											
Agrostis capillaris				*										*	
Agrostis stolonifera	*	*		*		*	*				*	*	*	*	
Artemisia absinthium															
Artemisia vulgaris			*								*				*
Capsella bursa pastoris											*				*
Cardamine hirsuta	*					*									
Cerastium fontanum						*									
Cirsium arvense															
Crepis capillaris								*							
Epilobium spp.	*			*	*	*		*	*	*	*	*		*	
Eupatoria cannabinum						*									
Holcus lanatus	*	*		*						*		*	*	*	
Hypericum perforatum															
Hypochaeris radicata															
Juncus bufonius					*										
Juncus spp.							*								
Linaria vulgaris															
Medicago lupulina						*									
Plantago major															
Plantago lanceolata						*									
Poa annua		*	*	*											*
Poa trivialis	*		*							*	*				
Ranunculus repens														*	
Rumex acetosella															
Rumex obtusifolius	*	*	*							*					
Sagina procumbens								*		*					
Senecio vulgaris						*									
Sisymbrium officinale			*												
Solidago spp.		*				*									
Sonchus spp.		*		*	*										
Taraxacum agg.														*	
Trifolium dubium															
Trifolium pratense															
Trifolium repens		*													
Tripleurospermum inodorum											*				*
Urtica dioica			*								*				
Vicia hirsuta															

***Table 4.3** (continued)*

	16) Sm Bentley Mill	17) Lg Bentley Mill	18) Hall Green	19) Sandy Lane	20) Vincent Drive	21) Saltwells	22) Cole Bank	23) Samson Quarry	24) Turners Hill	25) Burberry	26) Cole Round	27) Mid Cole	28) Bearwood R'way	29) Wilson Rd	TOTAL
Achillea millefolium															1
Agrostis capillaris		*											*		4
Agrostis stolonifera	*	*	*	*	*	*	*	*	*	*		*	*	*	22
Artemisia absinthium								*							1
Artemisia vulgaris		*	*					*	*			*		*	9
Capsella bursa pastoris							*								3
Cardamine hirsuta		*		*											4
Cerastium fontanum			*	*					*						4
Cirsium arvense												*			1
Crepis capillaris									*						2
Epilobium spp.	*	*	*				*	*	*	*	*	*		*	20
Eupatoria cannabinum															1
Holcus lanatus			*	*		*	*	*	*		*	*	*	*	17
Hypericum perforatum														*	1
Hypochaeris radicata													*		1
Juncus bufonius															1
Juncus spp.							*		*		*			*	5
Linaria vulgaris			*												1
Medicago lupulina	*								*			*			4
Plantago major	*								*						2
Plantago lanceolata		*	*		*					*	*	*		*	8
Poa annua									*		*				6
Poa trivialis		*										*		*	7
Ranunculus repens															1
Rumex acetosella								*							1
Rumex obtusifolius						*	*			*	*			*	9
Sagina procumbens	*		*				*		*		*			*	8
Senecio vulgaris															1
Sisymbrium officinale															1
Solidago spp.		*		*						*					5
Sonchus spp.															3
Taraxacum agg.															1
Trifolium dubium				*											1
Trifolium pratense										*					1
Trifolium repens															1
Tripleurospermum inodorum															2
Urtica dioica	*	*				*						*			6
Vicia hirsuta							*								1

Figure 4.3: *Detrended Correspondence Analysis (axis 1 vs. axis 2) for species composition of the seed bank and the overlying vegetation at 29 derelict land sites in the West Midlands.*

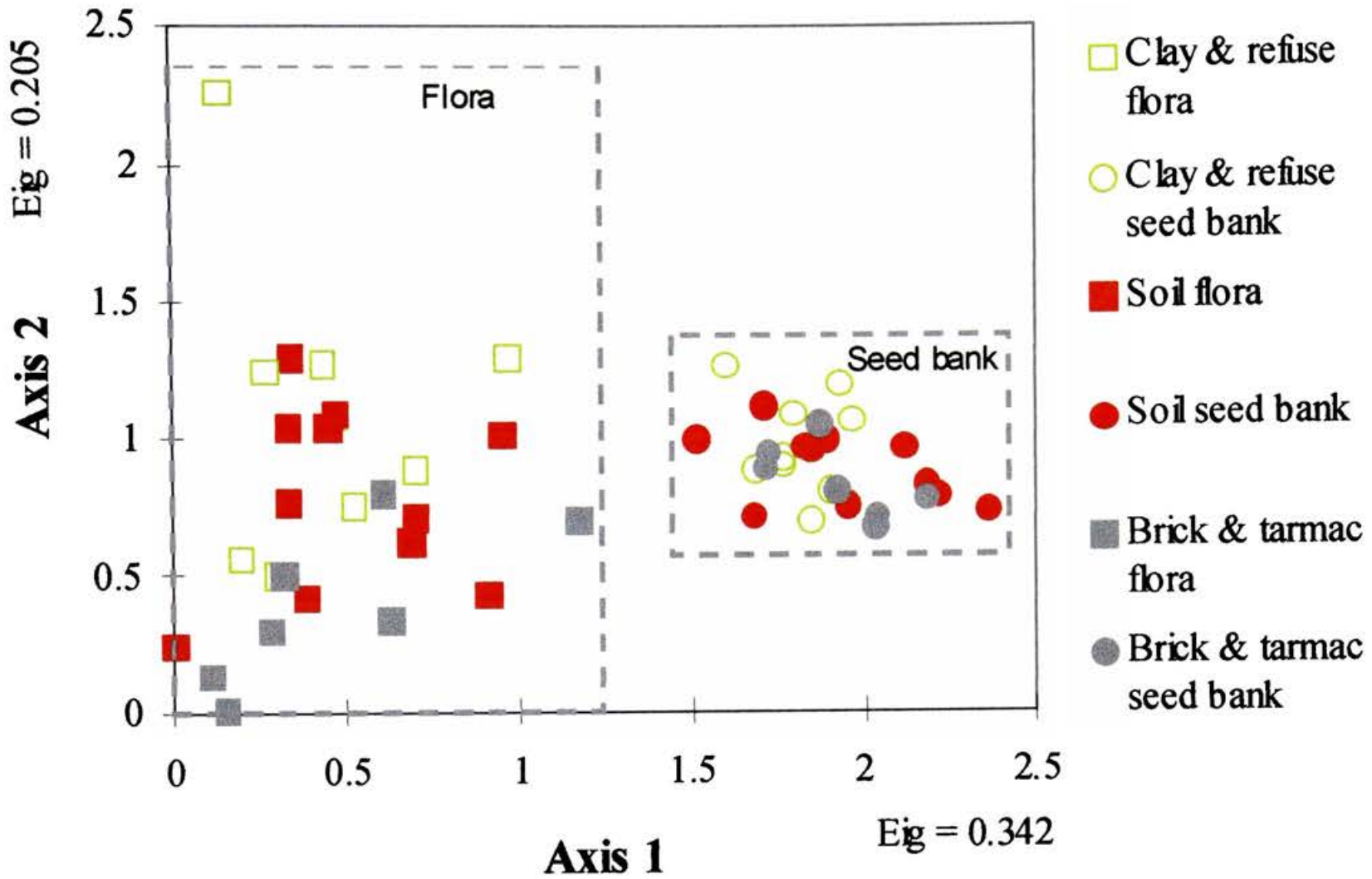
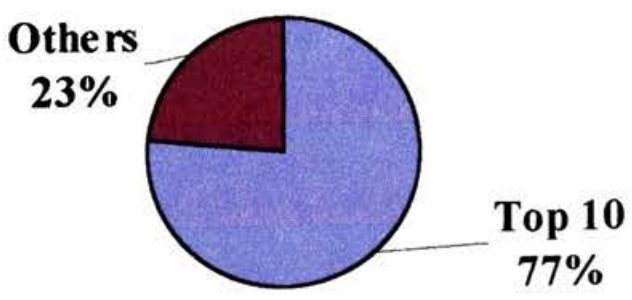
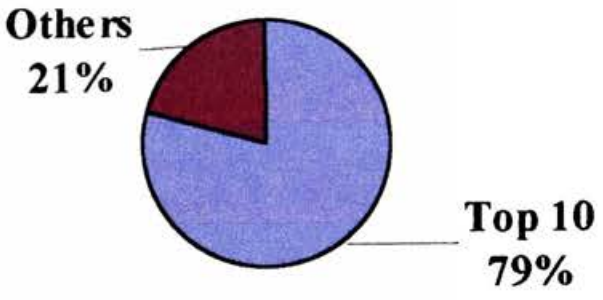


Figure 4.4: *The proportion of germinable seeds that are of the ten most abundant taxa found in the survey, where the major substrate is brick rubble, clay infill, refuse, or garden topsoil.*

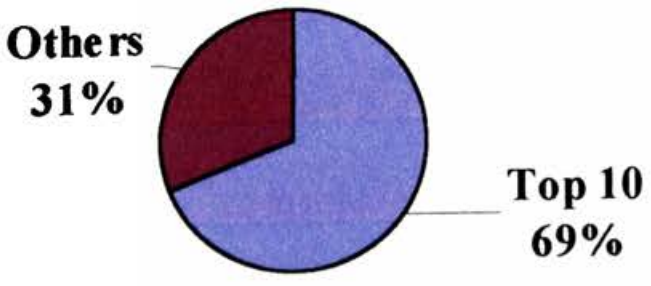
(a) Brick Rubble (5 sites)



(b) Clay infill (5 sites)



(c) Refuse (4 sites)



(d) Garden soil (6 sites)

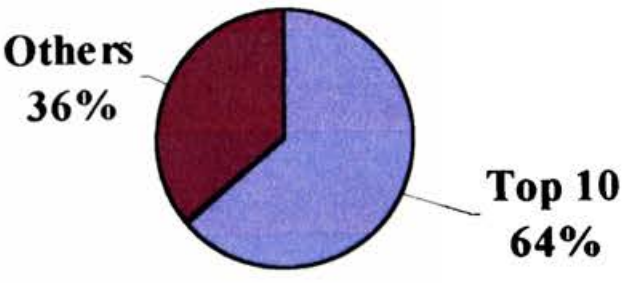


Table 4.4: *Ten most abundant taxa in the seedbank at survey sites*

TAXA	NO. OF SITES AT WHICH PRESENT	MEAN SEED DENSITY WHERE PRESENT (M^{-2})
Agrostis stolonifera	29	1,770
Epilobium spp.	29	2,822
Poa annua	28	505
Holcus lanatus	28	940
Rumex obtusifolius	25	493
Urtica dioica	25	380
Sagina procumbens	21	1,693
Juncus spp.	19	911
Artemisia vulgaris	19	1,231
Solidago spp.	12	1,540

Table 4.5: *Comparison of some plant characters between the most abundant species in the seed bank study, in the derelict flora and in the regional flora (* $p < 0.05$, ** $p < 0.01$, *** $p < 0.005$, **** $p < 0.001$) Regional flora data was provided by the Ecorecord database of the Birmingham and Black Country Wildlife Trust.*

CHARACTER	MEAN VALUE			T-TEST SIGNIFICANCE LEVELS		
	SEED BANK	DERELICT FLORA	REGIONAL FLORA	SEED BANK VS DERELICT FLORA	SEED BANK VS REGIONAL FLORA	DERELICT FLORA VS REGIONAL FLORA
Seed longevity	0.59	0.44	0.39	**	***	n.s
Log Seed Weight (mg)	-0.37	-0.26	0.05	n.s.	*	n.s.
Seed shape	0.09	0.10	0.10	n.s.	n.s.	n.s.
Log(Mx√Vs)	-0.86	-0.80	-0.72	n.s.	n.s.	n.s.
S.L.A. (sqrt)	5.20	5.07	4.72	n.s.	**	*
Dry matter	4.25	4.49	4.84	n.s.	****	*
Life span	N/a	N/a	N/a	*	****	***
Max plant height (cm)	139.8	191.0	563.7	n.s.	**	*

Figure 4.5: *Representation of longevity index values by the 50 most abundant species recorded in a) seed bank of derelict sites b) the associated derelict vegetation, and c) the regional flora for the West Midlands region. Regional flora data were provided by the Ecorecord database of the Birmingham and Black Country Wildlife Trust. (Appendix VIII)*

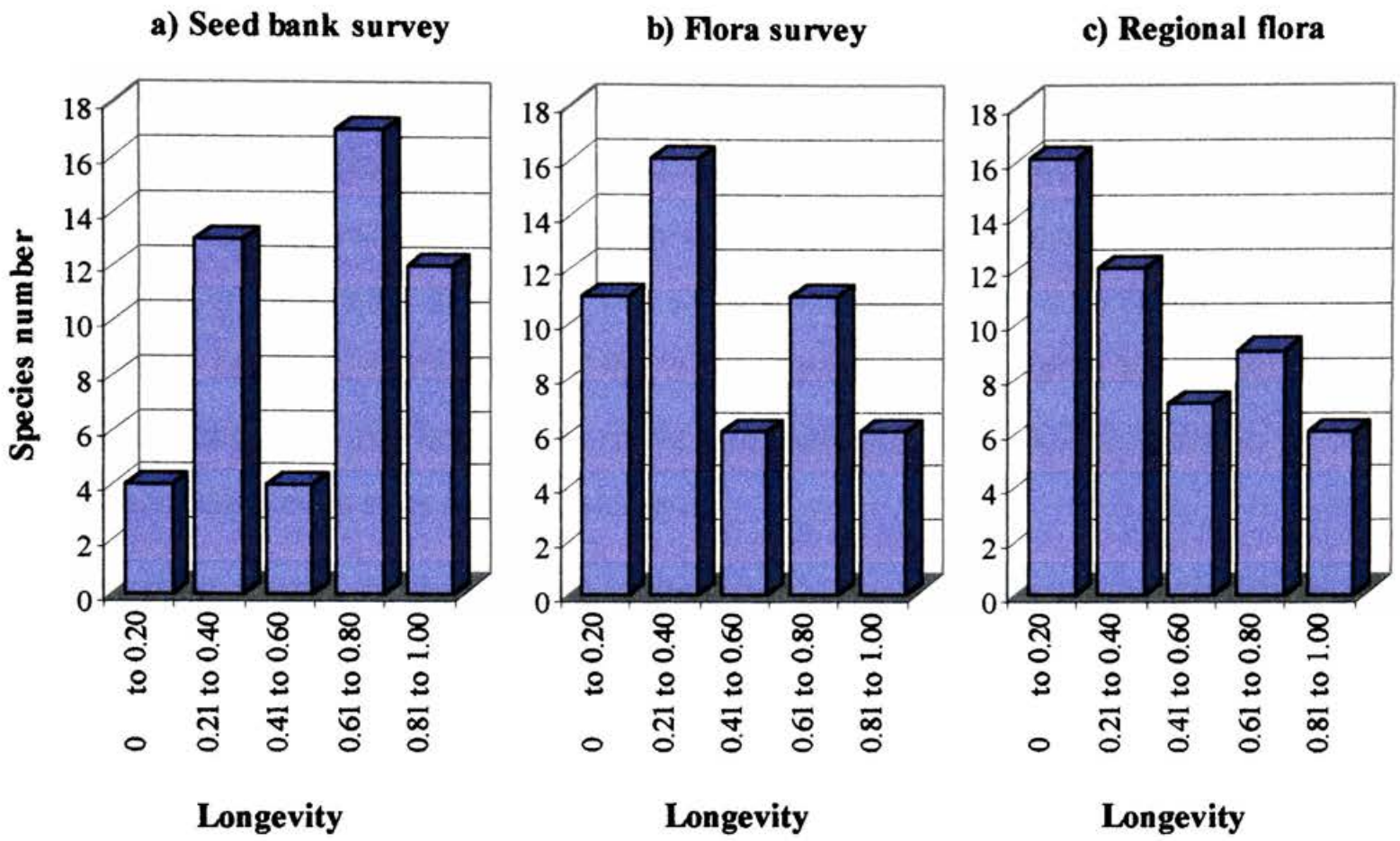
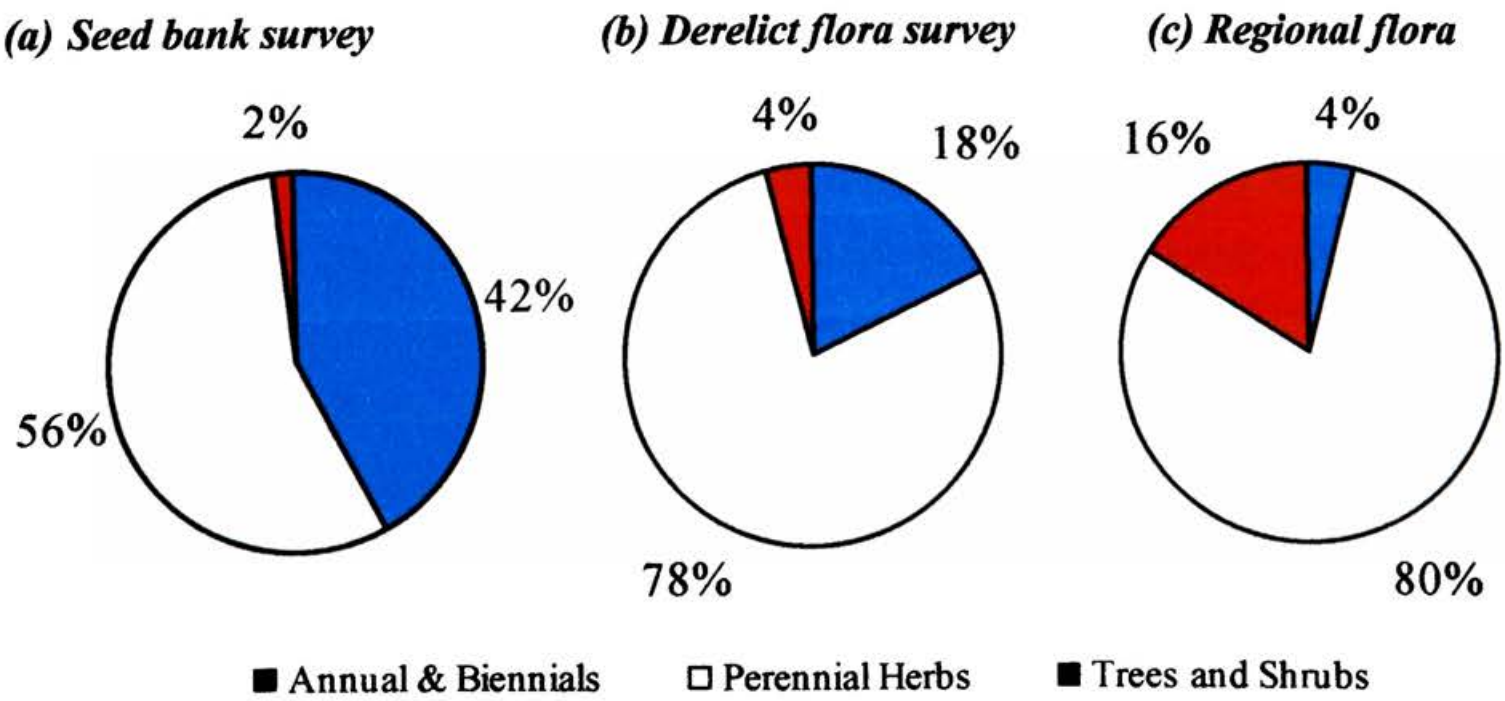


Figure 4.6: *Representation of life histories by the 50 most abundant species recorded in a) seed bank of derelict sites b) the associated derelict vegetation, and c) the regional flora for the West Midlands region. Regional flora data was provided by the Ecorecord database of the Birmingham and Black Country Wildlife Trust.*



4.4 RESULTS (2) DEPTH DISTRIBUTION IN THE SOIL

A comparison of germinable seed contents of 0-5cm and 5-10cm cores was carried out for species number, seed density and similarity with the vegetation (Figure 4.7a-c). Statistical comparison was carried out using Wilcoxon signed-ranks matched pairs test. At all eleven sites for which split cores, were taken both seed density and species number were found to be higher in the upper layer of soil. Similarity between the seed bank and the vegetation was calculated using Sørensen's index, and was also higher in the upper core at ten of the eleven sites. Patterns of depth distribution of the most abundant species were tested for expected changes in plant traits. All taxa found above a minimum density of 100 seeds m^{-2} (0-10cm depth) across the eleven sites were considered.

The proportion of total seedlings found in the top 0-5cm soil layer was then plotted for each of these species along the x-axis against a range of seed characters (Figure 4.8a-d). Species whose seeds decline most rapidly with increasing soil depth will have a high value on the x-axis. The Spearman rank method was used to test the correlation. The existing seed longevity index of Thompson, Bakker & Bekker (1997) proved to be the best predictor of depth distribution for seeds on the derelict sites (Figure 4.8a). The function Log (M x $\sqrt{Vs}$) also explained the depth gradient, although to a lesser extent (Figure 4.8b). However seed weight alone proved a better explanatory variable (Figure 4.8c), as there was no relationship between seed shape and depth distribution in this study (Figure 4.8d).

4.5 RESULTS (3) CHANGES IN SEED BANK COMPOSITION ASSOCIATED WITH SUCCESSION

The germinable seed bank (0-5cm depth) was compared for all sites in respect of their age since dereliction. As expected the similarity of the seed bank with the vegetation quadrats declined significantly along the successional gradient (Figure 4.9a). Seed density also declined with increasing age but this trend was not significant (Figure 4.9b). Changes in other plant trait data were tested by taking the mean values of certain characters of all the species found in the seed bank for each site and correlating with age. This showed a significant decline in specialism (Figure 4.9c) as calculated by the index of Thompson, Hodgson &

Figure 4.7: *Comparison of the seed bank found at different soil depths for a) seed density b) species number and c) similarity with the flora. Significance levels shown are for Wilcoxon Rank Sum tests.*

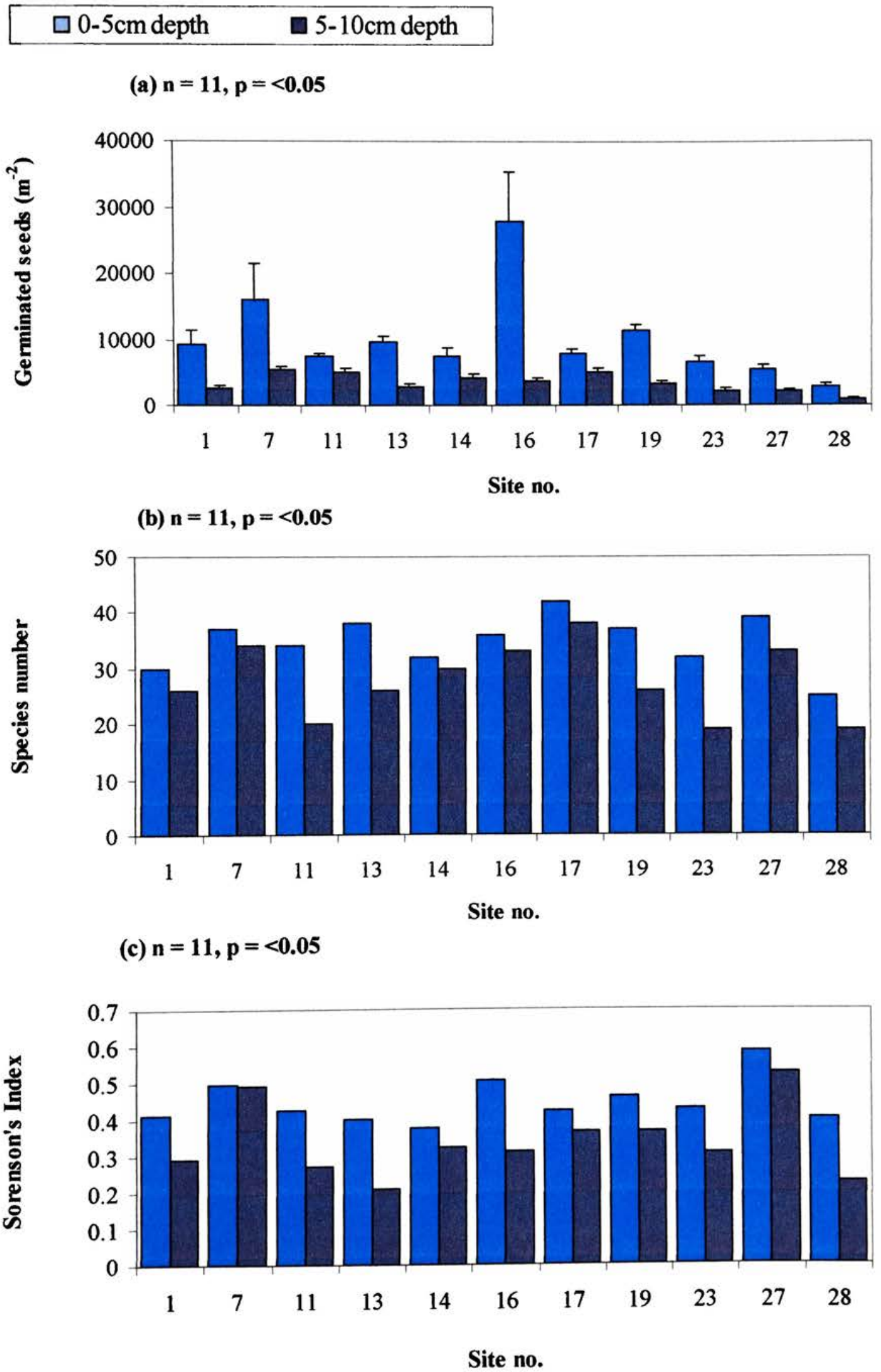


Figure 4.8: *Relative depth distribution of seeds of the most abundant taxa present in the seed bank at 11 sites studied with split cores showing the relationship with key plant characteristics a) seed persistence b) weight x shape c) weight, and d) shape..*

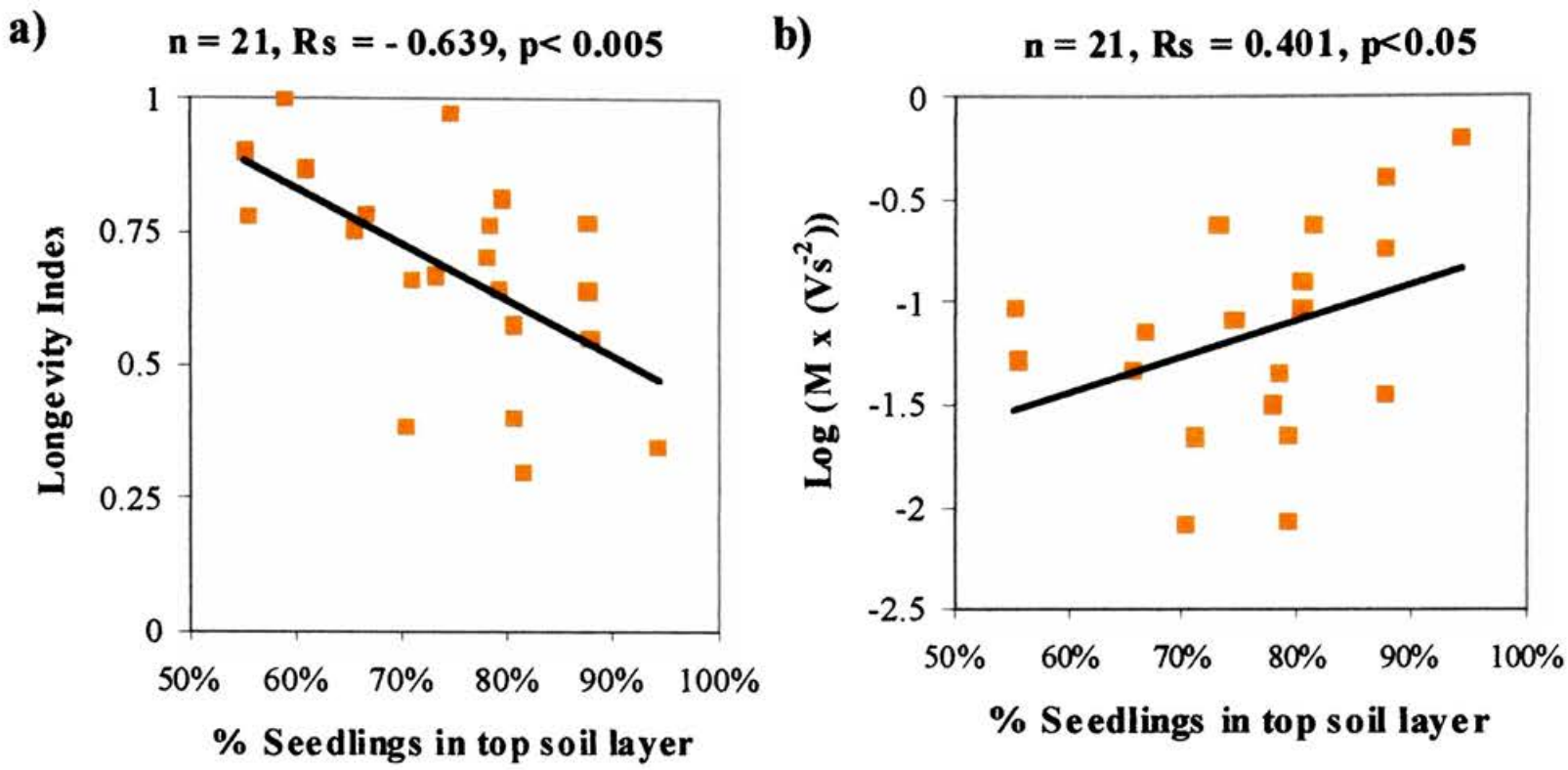


Figure 4.9: *Changes in the composition of the seed bank of derelict sites along a chronosequence a) similarity with the overlying vegetation and b) the density of germinable seeds recorded c) mean specialism index value, and d) seed persistence.*

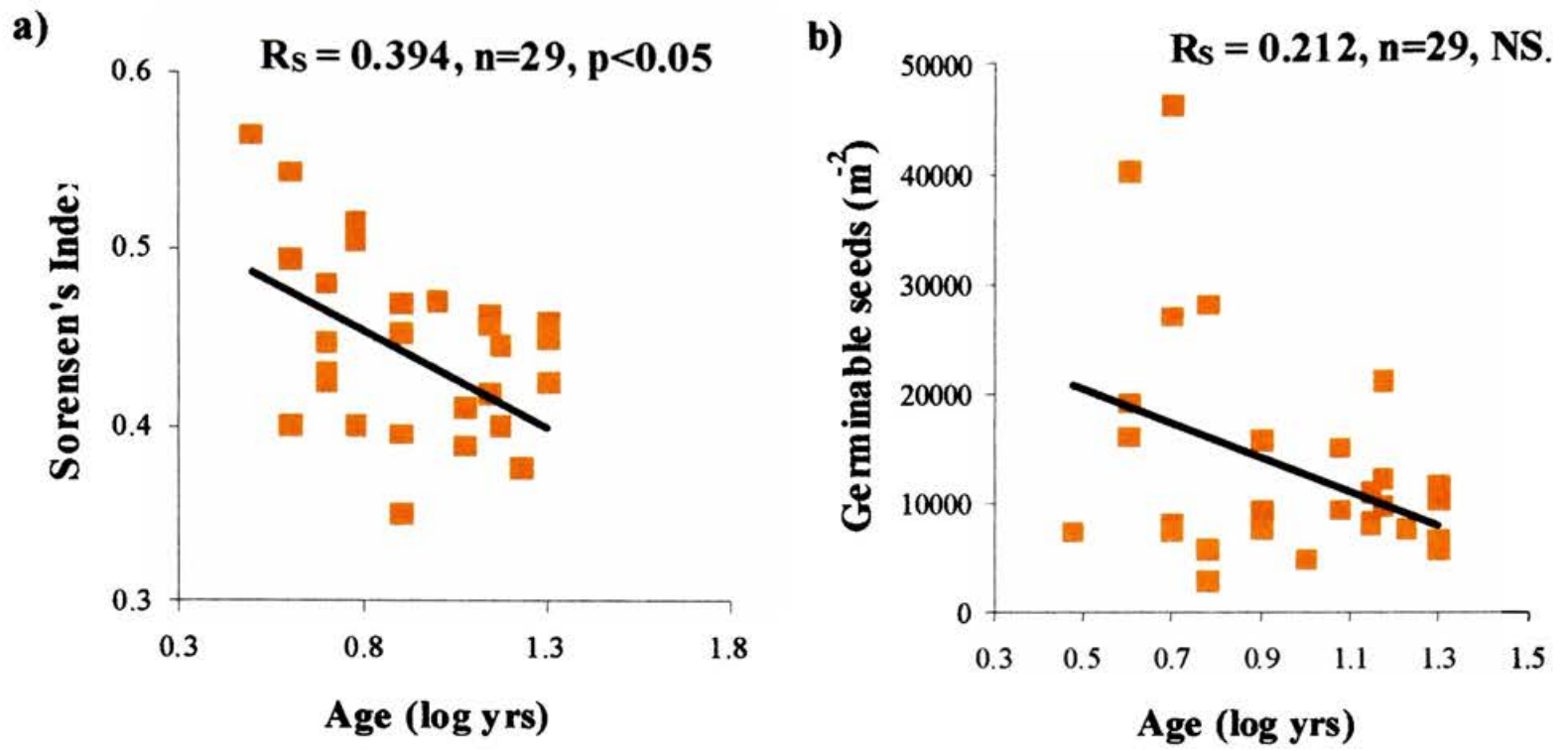
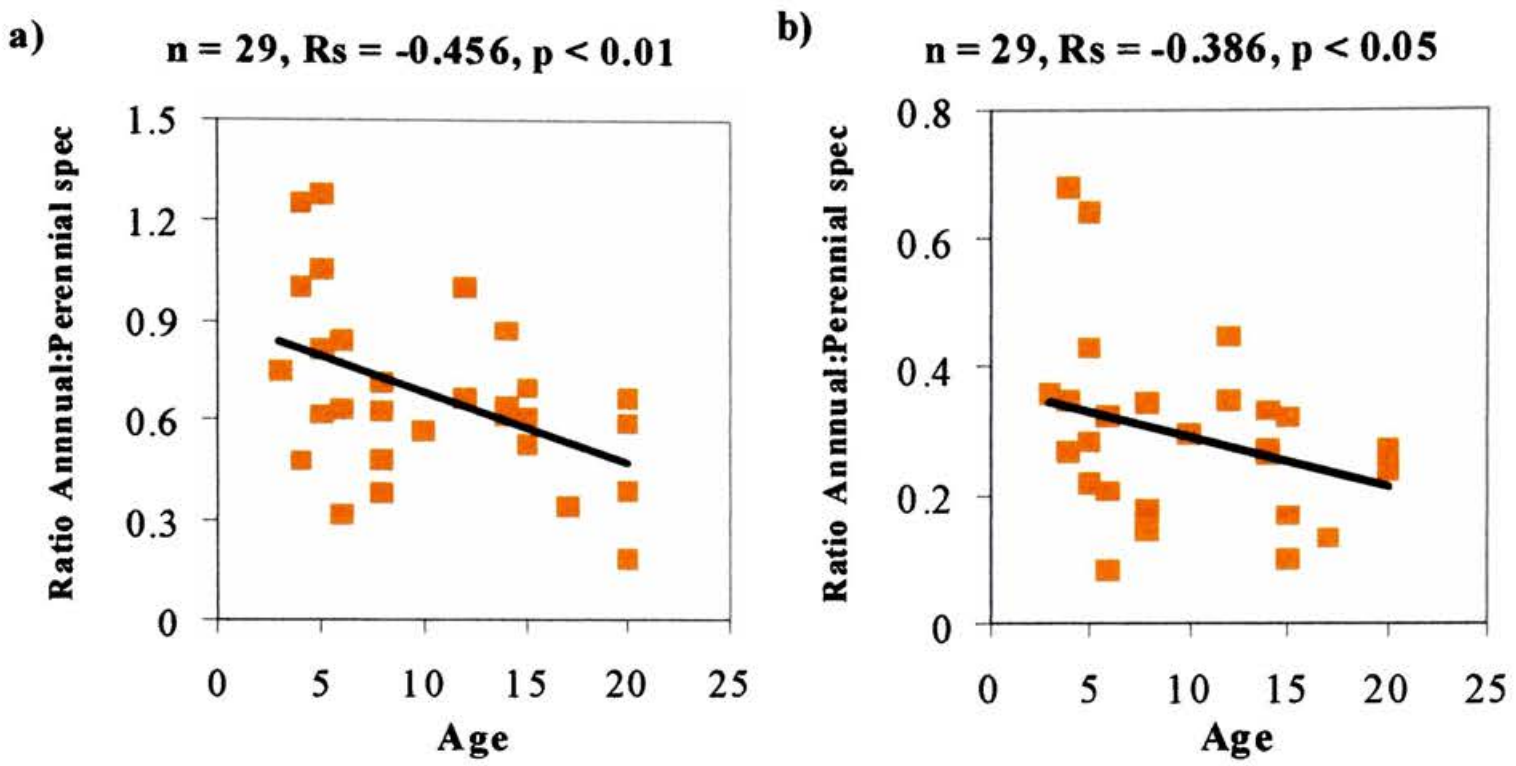


Figure 4.10: *Changes in species composition with increasing age on derelict site. Figs a & c are derived from seed bank data and Figs b & d are from the vegetation data. Figs a & b show change in the numbers of annual and perennial species. Figures c & d show change in the relative abundance of annuals and perennials.*



Gaston (1998b) for which increasing value indicates higher habitat fidelity. Mean seed longevity of species represented in the soil also declined (Figure 4.9d).

Figure 4.10a-d illustrates the change in strategy associated with species in the vegetation along the age gradient. In terms of species present, increasing plant longevity is significantly favoured as a site ages, reflected by a higher number of perennial species found in both the seed bank (Figure 4.10a) and the flora (Figure 4.10b). However, while this pattern is also reflected in terms of abundance in the flora (Figure 4.10d), the densities of annual and perennial seeds in the soil remains much more constant across the site age gradient (Figure 4.10c).

4.6 DISCUSSION

Review of methods

A relatively small amount of greenhouse space was required to give a realistic assessment of the seed densities of the common species at 29 different sites, providing a strong recommendation for Ter Heerdt *et al.*'s (1996) condensing and emergence methods. However, the very high seedling densities that can occur after condensing meant that during the peak period of germination the time commitment to identification and removal was high. To prevent overcrowding and seedling mortality during this study required, between 10-12 man hours daily for a period of about three weeks. After this period, the time input required significantly declined.

The additional germination period demonstrated that a majority of the viable germinable seeds in the soil had germinated after 14 weeks (mean = 94%). However, using the emergence method some of the results will be influenced by factors relating to the requirements for breaking of dormancy and are worthy of some discussion. Delayed germination of larger seeded legumes is probably a reflection of this group's requirement for scarification to allow water to imbibe. Smaller-seeded species probably received enough scarification during the sieving process to enable most to germinate. In contrast, larger-seeded species possibly benefitted more from the additional mixing which occurred at the end of the study period, especially if they came into contact with the more abrasive sand layer. By

examining the immediately germinable fraction of the seed bank, it is therefore possible that a number of larger-seeded legumes may be underrepresented. Indeed, *Cytisus scoparius,* which is commonly found at a number of urban sites, was only sparsely recorded in the seed bank. This species is known to have a persistent seed bank, undoubtedly a factor linked to its association with episodic fires (Grime, Hodgson & Hunt 1988). However, perhaps the most significant factor contributing to the low density of these large-seeded species in the soil is the trade-off between seed size and seed production capacity (Rees 1993).

Three other species stood out as having higher rates of germination in the additional period. *Cardamine hirsuta* is a winter annual, which requires high temperatures to break seed dormancy, and so additional levels may have been expected towards the end of the study as higher internal temperatures recorded, due to increasing light radiation levels from outside. Significantly, perhaps the drop in temperature at night would have been much less pronounced as the study progressed. *Hypericum perforatum* also has seed dormancy broken by high temperatures and may be explained similarly. *Veronica serpyllifolia* cannot be explained by this however, although it was only found at one site.

General features of derelict seed banks

Seed densities varied quite significantly between sites and possible reasons for this are discussed later. Nevertheless it is clear, as expected, that these sites have a relatively high density of seeds in their soil, considering their relatively recent origin and unstable nature. The number of species recorded for the study is also considerably high relative to those found in studies of more stable habitats e.g. woodland. This reflects the high diversity that was found in the flora. However it is noticeable from the findings that most of these species make quite insignificant contributions to the actual numbers of seeds found. The importance that the seed bank may have for maintenance of high diversity on these sites is therefore not clear.

It is evident that the dominant species are well-suited to forming well-distributed and abundant seed banks. Good dispersal ability, ability of seeds to persist long-term in the soil, high seed production rates, small seeds and ability to reach dominance in the vegetation are perhaps the most significant attributes. All of the ten most abundant taxa found in this seed bank study have at least one, and more often more, of these characters. Three are members of the Asteraceae, and typically for this group, are well-adapted for dispersal to newly created

sites, producing large numbers of small seeds that can become easily buried in the soil. *Epilobium* species also have seeds with plumed hairs aiding dispersal. Their seeds are also known to contain significant levels of ortho-dihydroxyphenol which is linked to high persistence (Hendry *et al.* 1994). High seed production capacity and very small light seeds are also the key to the abundance of *Juncus* species seeds found, despite their comparative rarity in the vegetation. *Rumex obtusifolius* has a toothed perianth facilitating animal dispersal and also has seeds capable of long term persistence (Grime, Hodgson & Hunt 1988). While *Urtica dioica* has no specialised adaptations to dispersal, it appears to be a very successful colonist either by seed or rhizome fragment and certainly, once established, it can produce large numbers of seeds from extensive clonal stands. *Agrostis stolonifera* is another species that has the ability to quickly colonise and become dominant and this is reflected by the high seed numbers it deposits. Of the other grasses, *Holcus lanatus* produces large numbers of well-dispersed seeds which are adept at colonising gaps and *Poa annua* has proven capable of creating a long-term buried seed population (Roberts & Feast 1972) and of taking advantage of gaps at any time of year, due to its capacity for all year round immediate germination.

The gradient of trait values from the regional flora through to the derelict seed bank can be closely related to current changes in the distribution and abundance of species in the British flora. Plant traits most strongly associated with species in the seed bank and flora of derelict sites, notably longevity of seeds and short life-span, are strategies which are proving successful within the modern landscape of Britain (Hodgson & Grime 1990). It is the combination of disturbance and high productivity which promotes these traits within these sites. However, the fact that the species in the seed bank represent a further step along this trait gradient from those in the flora emphasises the advantage that formation of a persistent seed bank carries.

Depth distribution

Lower seed densities in deeper soil fits the expected pattern, as vertical depth is assumed to reflect time since deposition. Rapid loss of seed viability of many transient species means that they are rarely represented beyond the uppermost few centimetres of the soil. Short-term persistent species are also likely to be lost as depth increases. However with time seed density can, under some circumstances, become higher in lower depths of soil (e.g. Grandin

& Rydin 1998). Most commonly, this is likely to occur where the climax community produces a considerably smaller and transient seed rain and where the previous successional series involved high deposition of long-term persistent species. Once buried at depth, germination in these seeds will be restricted by lack of light (Matlack & Good 1990). Such a finding was not to be expected at any of the sites in this study however because even at the oldest sites, deposition rates are high and many persistent seed strategists survive in the vegetation.

It has been acknowledged that studies of seed banks from a wider range of habitats would be beneficial for testing existing knowledge of persistence (Thompson *et al.* 1998a). This study is consistent with the view that the existing values of longevity derived from Thompson, Bakker & Bekker's (1997) database provide a sufficiently representative value for the common species found in the seed bank. Ultimately, however, the accumulation of more data from under-recorded habitats could potentially continue to provide refinements for some time to come. This will probably be most significantly the case for species on which the present index is derived from a small number of records only.

Despite the strong prediction provided for depth distribution by the longevity index, this study does provide some results that conflict with one predicted pattern. Seed shape does not appear to play much of a role in the segregation of seed numbers from the common species between upper and lower soil layers. Several studies since the development of the longevity index (e.g. Funes *et al.* 1999; Moles, Hodson & Webb 2000) have made similar observations. This could be explained by the regularity of anthropogenic disturbance events and, in particular, tipping of garden refuse, that can replace natural mechanisms of burial into the soil. Artificial forms of burial can give an illusory impression of true persistence which are not strictly appropriate (e.g. Pons 1989). It is consequently fair to assume that the consideration of size and shape to predict depth distribution is perhaps most likely to find difficulties on sites such as these, where the types and intensity of disturbances are themselves so unpredictable.

Age

While declining similarity between seed bank and vegetation along a successional gradient has been found before (Donelan & Thompson 1980; Roberts & Vankat 1991), it was perhaps unexpected that such a clear trend would be exhibited along the site age series presented here.

In contrast to these previous studies, the age gradient considered here is far shorter, representing only about 20 years and the type of sites, though by definition derelict, are quite variable. Furthermore, the regularity of disturbance events might be expected to have increased similarity on older sites, as they offer the opportunity for recolonisation by pioneers retained in the seed bank but which would normally have been suppressed in the vegetation. The decline in seed density with age also fits the expected patterns but it is notable that the gradient is less steep than for similarity. It is possible that small-scale disturbances may explain this difference. Events like small fires, often started on these sites by local youths, create small gaps for early successional species to regain a foothold, either from the existing seed bank or from the seed rain. However, the duration for which they remain in the vegetation under these circumstances is likely to be much shorter, when compared with their presence on arrival after denudation. This is because there is already a surrounding source of competitive species, particularly perennial grasses, which can rapidly invade the gap vegetatively. This is evident to anybody who has walked over urban commons and witnessed the manner in which fire patches quickly disappear so that their position is often unrecognisable from the rest of the vegetation within a year or two. By this point the similarity of the vegetation with the seed bank, briefly increased after the gap formation, will fall back again. However, in the time they are present, the pioneer species, generally typified by high reproductive capacity, may have deposited high numbers of seed around the site. It is possible under this scenario for similarity with the flora to decline along the successional gradient, with some degree of oscillation, while seed density remains high. Indeed in support of this theory it is notable that the older site which deviates most from the pattern of declining density is Vincent Drive where a range of small scale disturbances including small fires and motorbike racing occur sporadically throughout the year. However, it is likely that substrate type is another contributory factor to anomalies. Where comparatively low levels of seed density were found at younger sites, the soil was typically a heavy clay which may, perhaps, be less penetrable for seeds, consequently making them more vulnerable to predation on the surface.

The increasing representation of perennials in the vegetation of the sites along the successional series, fits expected patterns of change in life history. Species with higher longevity were found in both increased numbers and increasing dominance in later

successional series, as demonstrated by previous studies (Clements 1916; Egler 1954). In contrast, the seed bank continues to contain high density of annual seeds at older sites, even though plant life span increases. Clearly, while the increased numbers of perennial species in the soil reflects changing deposition from the vegetation above, a combination of long-term persistence and higher seed production maintains the greater numbers of pioneer annual seeds. Pioneer species are strategists adept at benefiting from the ephemeral nature of derelict sites. Change associated with the ageing of sites is the key to invasion by a wider pool of generalists. This change may be linked either to the environment of the habitat patch itself, or may simply relate to time taken for sufficient colonising material to reach the site. For the pioneer species, their subsequent disappearance from the vegetation is mitigated by their continuing long-term persistence in the soil, while awaiting the next disturbance event.

Further contributions to the understanding of urban seed banks

While this study provides a basis for understanding the nature of seed banks in the soil under derelict urban land, further research could undertake clarify some of the cloudier issues. A more detailed study of the spatial patterns of depth distribution across a site could illuminate the significance of artificial burial due to tipping. The nature of the regeneration communities found after different types of small-scale disturbances and how they relate to the seed in the soil also offers scope for further study.

4.7 CONCLUSIONS

a) In general the patterns of distribution for germinable seeds found in the soil of derelict sites closely matched the expected predictions.

b) Seed banks on derelict land have relatively high densities of seed in the soil as was to be expected for unstable sites of recent origin;

c) The seed banks of different sites are very similar, even where the overlying vegetation differs considerably.

d) Similarity with the vegetation declines significantly with both increasing depth in the soil and as sites progress along a secondary succession. This is in agreement with the findings of previous studies along successional gradients of different types. The

density of germinable seeds recorded also declines deeper into the soil and at older sites, though not significantly so for the latter;

e) The longevity index derived from studies on a range of other habitats provides a good prediction for relative distribution of seeds at different depths on derelict habitats. Seed weight is also important in determining whether seed is found deeper in the soil. Lighter seeds are more easily incorporated to greater depths. However seed shape did not prove a significant factor against the expectations;

f) It is postulated that the considerable anthropogenic influence on derelict land sites may account for some unexpected findings. Small-scale disturbances of differing intensity and frequency could potentially influence burial rates and/or change the impact of successional progression on the seed content of the soil.

CHAPTER FIVE

URBAN WILDLIFE CORRIDORS – LANDSCAPE MANAGEMENT TO PRESERVE NATIVE PLANTS?

SUMMARY

Urban planners increasingly have to consider the long term impact of development schemes on the wildlife of a city. In the last decade, policy guidelines have frequently highlighted the importance of interconnected green spaces for maintaining diversity in cities. The conservation, restoration and creation of linear habitat remnants, termed 'wildlife corridors' has been actively encouraged to enhance connectivity.

Considerable investment has been placed into the designation and creation of wildlife corridors in this period despite scientific evidence to support their utility being inconclusive. In this review the findings from recent studies of plant dispersal along linear features and the implications of these for conservation in cities are considered. Also discussed is the theoretical validity of applying wildlife corridors within an urban context.

It is argued here that linear features in cities can be placed into two major groups. The first group, urban greenways, are mainly heterogeneous linear belts of woodland, grassland and riparian communities with an established recreational value. The second group, termed here as greyways, are discrete narrow homogeneous linear features such as walls, pavements and road verges containing early successional communities which are densely interconnected across a city.

It is suggested that the predominant use of the term wildlife corridor by planners for urban greenways is a misapplication of the concept as these features invariably do not provide habitat connectivity. Furthermore the mobility and ecological requirements of most native species found within them is not compatible with movement across such a

heterogeneous landscape. Evidence suggests that their ability to function as corridors is low for most native species.

In contrast, greyways provide a higher level of habitat connectivity and are occupied by species capable of rapid dispersal. Evidence suggests their ability to function as wildlife corridors for native species is much higher.

The evidence from research shows that many alien species are well suited for dispersal along linear features. While greyways will tend to contain more aliens which move faster, the spread of aliens along greenways is potentially more threatening to sensitive habitats, as species capable of dispersing along them are, by definition, better suited to proliferate in stable natural and semi-natural habitats.

KEYWORDS: *Urban ecology, Cities, Landscape, Wildlife corridors, Plants, Conservation*

5.1 INTRODUCTION

The fragmentation and isolation of natural and semi-natural habitats is an issue of increasing concern to conservationists and has been identified as a major contemporary threat to biodiversity (e.g. Harris 1984; Wilson 1988, Saunders & Hobbs 1991; Pickett *et al.* 1997; Hess & Fischer 2001). A popular tool to mitigate habitat fragmentation is the generic 'wildlife corridor'. This is reflected in local, national and international initiatives and policies. Agenda 21 of the Rio Earth Summit Convention requires the member signatories to ensure the conservation of biodiversity at a local level. Within the UK, the responsibility for implementing policy designed to fulfil this obligation rests largely with local government. In recent years, policy has been particularly directed by the European Habitat and Species Directive's (Planning Policy Guidance Note. 9) recommendations for European member States. This states that they 'shall endeavour in their land-use planning and development policies to encourage the management of features of the landscape which are of major importance for wild fauna and flora. Such features are those which, by virtue of their linear and continuous structure or their function as stepping stones, are essential for the migration, dispersal and genetic exchange of wild species.' Policy at local government level increasingly

reflects this directive, placing great importance on the creation and protection of wildlife corridors or urban greenways to mitigate the fragmentation and isolation of habitats and populations (*e.g.* Greater London Council 1984; Land Care Associates 1997).

Impacts of habitat fragmentation

Within the UK, large areas of the natural and semi-natural vegetation of the landscape have been removed, leaving only scattered remnants, distributed as isolated patches or as linear features. Two of the most prolific modern landscapes of lowland Britain, agricultural and urban, exhibit this high intensity fragmentation of natural and semi-natural habitats (Spellerberg & Gaywood 1993). Remnant fragments within the agricultural landscape are isolated within a matrix of predominantly intensively farmed fields, while remnant fragments in the urban landscape are isolated by a more complex matrix of development, including business, residences, industry and transport infrastructure.

The detrimental effects to biodiversity that are likely to occur when a natural or semi-natural landscape becomes fragmented have been well documented in ecological theory. Theories of island biogeography (MacArthur & Wilson 1967) and metapopulation dynamics (Levins 1969) have become the basis for many contemporary approaches to conservation biology and landscape management, including the management of fragmented ecosystems. Island biogeography uses island size and rates of immigration and extinction to predict the number of species on an island. Metapopulation theory uses intra-population dynamics and inter-population movement to describe the population dynamics of individuals of a species existing in a pattern of habitats patches. Much of the experimental work has focused on fauna, but the decline in species richness predicted for fragmenting systems (Terborgh 1976; Wilcox 1980) has been demonstrated for plant species on chaparral vegetation (Soule *et al.* 1992), for calcareous grasslands (Fischer & Stocklin 1997) and within the UK in woodland (Bastin 1997). This loss of botanical diversity has been shown to be linked to increasing distance between patches reducing the probability of immigration and inter-patch dispersal (Primack & Miao 1992). Habitat generalists have been found to be less vulnerable to extinction than habitat specialists (Fischer & Stocklin 1997). The ability to persist in adjacent patches of variable habitat type and quality will be advantageous to recolonisation after local extinction events. Specialist taxa with close relationships with animals for either pollination (Rathcke &

Jules 1993) or for seed dispersal (GrashofBokdam 1997) are more sensitive to declining connectivity. The risk posed to these species will ultimately always be closely linked to their respective vectors' response to changing habitat availability. Consequently, for a species to succeed within a fragmented system, it pays to have a flexible strategy, and generalist species that have efficient dispersal mechanisms will be at a competitive advantage.

5.2 THE WILDLIFE CORRIDOR CONCEPT

The idea that connecting linear habitat strips could ameliorate the negative effects of fragmentation became well established towards the end of the last century (Diamond 1975; Wittig & Schreiber 1983; Noss 1983; Merriam 1984; Forman & Godron 1984,1986) and is rooted in both island biogeography and metapopulation theory. Corridors may enhance the movement of species between 'island' habitat patches, increasing immigration and therefore species number according to island biogeography theory (Willis 1974; Wilson & Willis 1975). Likewise corridors may enhance the movement of individuals between habitat patches, reduce the time for recolonisation of patches which have suffered extinction, and facilitate the action of 'source' populations – well populated patches which produce emigrants to 'rescue' patches with depleted populations (Brown & Kodric-Brown 1977). A review of reasons advocated for supporting linked wildlife corridors has been prepared by (Dawson 1994a) and includes their utility in response to global warming.

Within the urban context considerable importance has been attributed to corridor features in the last ten years, although the significance of linear habitat fragments has previously been recognised (see e.g. Kelcey 1975b; Greater London Council 1984). In Essex, the county council targets the provision of wildlife corridors to be incorporated into all future urban development plans (Essex County Council 1998). In Birmingham, key corridors are already designated on the basis of both feedback from wildlife surveys and their accessibility to the public (Land Care Associates (LCA) Ltd 1997). The 'wildlife corridor' concept now forms the most clearly defined and widespread policy initiative implemented at local government level in the UK to maintain or enhance biodiversity. However the definition of the term 'wildlife corridor' is less clear, being used in ways that are contradictory and confusing (Table 5.1). Indeed it has been argued that, because corridors have different functions at

different spatial scales, and because the term is used by a range of disciplines (ecology, conservation, planning, landscape ecology, landscape architecture), a succinct and precise definition is neither possible nor desirable (Hess & Fischer 2001). With the rise of landscape ecology, the term has evolved from a functional description of a feature through which wildlife moves to a structural description of a linear feature that differs from the matrix on either side. 'Almost any strip of vegetation could be viewed as a corridor in some contexts' (Hobbs 1992, p.389). In accepting the structural definition of a corridor that is useful for landscape planning, it is useful to examine the functional aims of any planned corridor so that its effectiveness can be evaluated. Functional attributes of urban wildlife corridors include the ecological functions of habitat, conduit, barrier, filter, source and sink (Hess & Fischer 2001), and the social functions of recreation, ascetics and community cohesion (Forman 1995).

Table 5.1: *Functional attributes identified for wildlife corridors (after Hess & Fischer 2001).*

DEFINITIONS OF WILDLIFE CORRIDORS (Simberloff *et al.* 1992)
Distinct habitat, irrespective of whether it aids movement
Urban greenbelts and buffer zones
Biogeographic land bridges
Discrete 'stepping stone' refuges e.g. for wildfowl
Local wildlife passages e.g. road underpasses or tunnels
Strips of land facilitating movement between large habitats
FUNCTIONS OF WILDLIFE CORRIDORS (Andrews 1993)
Permit colonisation of new suitable sites
Allow emigration from changing and unsuitable sites
Permit recolonisation of sites where populations become locally extinct
Allow movement between different habitats needed for a species' life cycle
Increase total habitat extent
SOCIETAL GOALS OF WILDLIFE CORRIDORS (Forman 1995)
Protection of biodiversity
Enhancement of water resource quality and management
Enhancement of agroforestry
Recreation
Community and cultural asset
Dispersal route for species

(i) Habitat:

A habitat can be defined as a patch which contains the appropriate combination of resources and environmental conditions to allow the survival and reproduction of a species. Linear habitat fragments may be significant in their own right as refuge habitats in an otherwise unfavourable landscape.

(ii) Conduit:

There is confusion in the literature between the functions of conduit and habitat. If an area provides suitable habitat for a resident population, there is also a possibility that it may also function as a movement corridor or source area facilitating that species' emigration and dispersal in the area. However the strictest definition of a conduit is an area that provides for movement but not reproduction of the species. Any area that also provides for reproduction would, under this definition, be habitat (Rosenburg *et al.* 1995). This definition precludes functioning conduits that a species takes several generations to travel along, and may not be helpful when considering corridors for vegetation. Therefore a functioning botanical conduit of any significant length is likely to also be habitat.

(iii) Barrier and filter:

Filters imply selective permeability and reflect the fact that corridor function is species-specific. As species have unique habitat requirements, a wildlife corridor may perform different functions for different species. For some species, it may be unsuitable or disturbed habitat which they do not enter and thus may provide a landscape barrier to movement. A common example is a road which may be a conduit for people but a barrier to wildlife (Forman & Hesperger 1996). Corridors may thus filter out some species moving across or along them (Forman & Godron 1986; Thorne 1993), whilst acting as a conduit for other species (Vermeulen 1994).

(iv) Source and sink:

The terms source and sink are used in a demographic sense, sources being where local reproduction is greater than mortality, providing a pool of individuals for emigration, and sinks being where mortality exceeds reproduction so that the site relies on immigration to remain populated (Shmida & Ellner 1984; Pulliam 1988). Poorly designed corridors may act

as population sinks if the habitat is inferior or competition from edge and generalist species is high (Henein & Merriam; Soule 1991). Corridors which provide good habitat may act as sources for plant species to colonise adjacent areas (Thorne 1993). This may include invasive alien species as well as native species (Noss 1987).

The structural definition of a corridor is constant, and a useful definition for planning and surveying. The functions are specific, and include both ecological and social expectations. In terms of ecological aims, corridor functions will vary between species for any given corridor. Of these functions, habitat provision can be supplied as readily by an isolated site as by a wildlife corridor. Habitat provision alone therefore cannot rightly account for these interconnected habitats being conserved at the expense of potentially more biodiverse but isolated sites. If the implementation, establishment and maintenance of wildlife corridors within urban areas is to be considered cost-effective, and indeed, advisable at all on grounds of biodiversity, it is necessary to answer the questions of (i) does dispersal occur along urban corridors?; (ii) what species are likely to be successful dispersers?; and (iii) will the success of these species be beneficial to the biodiversity of the urban area as a whole? If dispersal is not occurring, or, if the species that disperse do not contribute beneficially to urban biodiversity, then the reasoning behind the preservation of urban linear habitat fragments as ecological corridors is flawed and there is a need to re-evaluate each fragment independently on the basis of the conservation importance of its component species or communities.

5.3 URBAN WILDLIFE CORRIDORS

The composition of habitat types and the physiognomic or spatial arrangement of those habitats are the two essential features which define a landscape (Dunning *et al.* 1992). Although a structural definition of urban wildlife corridors is commonly used, it is essential to be able to describe the broader significance of its distinct form upon the potential for species dispersal to understand the ecological processes occurring across the urban landscape. Understanding these factors may help predict which species are more likely to successfully disperse along urban corridors.

Most linear features classed as wildlife corridors in the urban landscape can loosely be grouped into two categories comprising of (a) large 'green' tracts of habitats, often with good

access to, and well utilised by, the public and (b) narrow 'discrete' strips of habitat, inaccessible to and/or undesirable for use by the public. The first category, referred to here as greenways, often incorporate large areas of the city's recreational greenspace. Frequently they may be linked to stretches of remnant habitats along water courses. Greenways may also be former transport links such as disused railways and their embankments. These linear features tend to contain stable, older communities such as secondary woodlands, scrub or mown coarse grasslands. While these corridors are large and visually impressive, being structurally distinct from the surrounding matrix, they are frequently made up of a mosaic of different habitat types and tend to be scattered across the city. The second category of corridor is a broad mix of discrete habitats referred to here as 'greyways'. These corridors are associated more with locations subject to often intensive daily human activity. Linear strips of vegetation of this type are found along road verges, railtrack, pavements and walls. Due to high levels of disturbance or recent origin, the communities found on these strips are usually dominated either by ruderal species or by other species capable of flourishing within a highly modified environment. Greyways, particularly walls and pavements, are most typically narrow but are commonly densely interconnected across any particular city.

5.4 URBAN GREENWAYS

Within the UK, designated urban wildlife corridors are often a habitat mosaic (see Plate 5.1) which may include semi-improved grasslands, scrub and secondary woodland but are often dominated by large tracts of mown playing fields and managed parkland. These interlinked green spaces have often survived planning pressure because they are adjacent to urban rivers or streams. These wildlife corridors are therefore much more analogous to the urban greenway concepts (e.g. Adams and Leedy 1987) of North America than to early descriptions of corridors which were applied to primarily homogenous linear habitat strips linking areas of pristine habitat. No two greenways are identical, and even where the major habitat type is the same, the differences can be stark when viewed on the ground. Width is rarely constant along the corridor, but varies widely with significant 'narrows' or 'pinch-points'. Gaps or barriers, such as bridges, also occur along greenways at differing frequencies and sizes. Current knowledge about how these discontinuities impair corridor function is poor (Dawson 1994),

Plate 5.1: *The River Cole, Birmingham City Council's "flagship" urban wildlife corridor*

(a) The R. Cole passing through the east of the city suburbs. Note the mosaic of woodland, scrub, rough grassland, recreational grassland and lake margin habitats along its route.

(b): The River Cole as it flow out of the urban zone at the very east of the city

***Plate 5.1** (continued):*

(c) A closer aerial view of River Cole corridor. The area of land to the south of the river is mown playing field with lawnmower lines still visible.

(d) An aerial view of the river Cole corridor illustrating the presence of road bridges, potential barriers to the movement of species.

but, as with width, the impact is likely be very dependent on the species concerned. The highly variable nature of urban greenways also poses a problem for their identification. An attempt at overcoming some of these difficulties associated with corridor designation was made by the City of Salford (Tomlinson 1995). Their method of selection was based on quantifiable measures of habitat volume, maturity, freedom from disturbance and ecological quality.

Urban greenways in the U.S. were envisaged, at least originally, primarily to improve recreational access and the environment of urban dwellers. Their value from this perspective has been comprehensively reviewed (e.g. Adams & Leedy 1987, Burgess *et al.* 1988, Barker 1995) and their preservation on this basis is not a matter for debate here. Recently much greater emphasis has been placed in the UK on the ability of these features to aid conservation. However it is important to recognise they do not provide true connections between remnant patches because many species cannot utilise, nor realistically disperse along, the intervening habitats. Consequently they cannot be valued or adopted validly on the same basis as true homogenous corridors as they certainly do not offer complete habitat continuity. There must therefore, instead be some consideration of the relative ability of species within these linked mosaic systems to traverse between patches. If the successful vagrants are vulnerable specialist species, then urban corridors could be important components of the landscape for conservation. However, if they encourage the transfer of mainly generalist species, then they could have the deleterious effect of giving these generally common species a competitive advantage. Indeed, an even worse scenario could be envisaged if invasive alien species are the major benefactors.

Plant dispersal and establishment in urban greenways

Zoochorous species may benefit from potential dispersal by mammals utilising riparian corridors in cities. Harris (1984) found that wooded sections of riverbank provided access for roe deer to access into a heavily urbanised area of Bristol, and grey squirrels have been found to utilise tree-lined stretches of river to move between larger wooded stretches (Gurnell 1987). In a Polish study, Goszcynski (1979) found that where urban river banks had not been faced with concrete a greater number mammal species could be found in the city. However urban rivers are generally highly modified linear features designed to cope with the high run-

off associated with impervious concrete and tarmac surfaces. The combined effects of large influxes of surface water and channelisation lead to much higher rates of flow than in unmodified rivers and, despite a more enlightened recent approach to river management, natural features such as islands, sand bars, pools and riffles have invariably been destroyed (Gilbert 1989). These factors and the poor water quality act to often make the rivers inhospitable to all but the most successful urban species.

Similarly, the regulation of urban rivers also prohibits against the deposition of seeds carried in the flow and they are consequently disadvantageous to dispersal of many hydrochorous species. In contrast, under natural conditions, hydrochory will have an important effect in structuring the communities found along the river and the frequency of species found would normally have a positive correlation with floating capacity. Only under these circumstances are continuous river corridors are likely to be important for maintaining regional biodiversity (Johansson *et al.* 1996). Within urban rivers, species richness is typically lower and the percentage cover of vegetation was significantly lower and the proportion of annuals is higher (Nilsson *et al.* 1991). The suggestion is that urban riverbanks are more open and that weedy early successional species are likely to persist longer beside urban river corridors than they would in more natural circumstances.

As greenways are regularly used for recreational purposes or as routes of travel by city dwellers the opportunities for the unwitting dispersal of plant seeds by people (anthropochory) are numerous. Dispersal by humans may occur directly by carrying seeds within mud on clothing (Salisbury 1964; Clifford 1956). Seeds might also be transported along recreational corridors in bicycle tyres or on the fur of dogs being exercised along the route. Grass seeds seem particularly well suited for dispersal within footwear (Clifford 1956). Notably however, Elton (1958) also considered that dispersal in footwear had played an important role in the spread of alien species.

There are also important indirect human impacts on the relative success of plant species in greenways. Under heavy human pressure, the microbial composition of the soil can be significantly altered (Zabinski & Gannon 1997) and the diversity of plant species has been shown to decrease with increasing compaction (Gomez-Limon & DeLucio 1995). A study of 18 vegetation types in the US (Cole 1995) found that the most sensitive species were the

chamaephytes while caespitose, matted and rosette hemicrypophytes and geophytes were most tolerant to the effects of trampling. Reproductive success of several tree species was also found to be negatively affected by recreational pressure in an urban forest park in New York (Loeb 1992).

Studies of native plant dispersal along river corridors are few, but the importance of these habitats as routes for the spread of invasive alien taxa has been repeatedly suggested (Pysek & Prach 1993, 1994; Cronk & Fuller 1995). The spread of several significant problem species has been recorded river banks including *Tamarix* spp in the US (e.g. Cleverly *et al.* 1997; DiTomaso 1998) and Australia (Griffin *et al.* 1989) and *Heracleum mantegazzanium* (e.g. Pyšek 1991; Kolbeck *et al.* 1994; Caffrey 1994) and *Impatiens glandulifera* (e.g. Perrins *et al.* 1993; Wadsworth 1997; Prowse 1998) within European catchments (Plate 5.5). Several factors may act to accelerate the spread of aliens along urban rivers. Native species are known to succeed at the expense of aliens when flooding is frequent (Pyle 1995) but within these regulated systems flooding is rare. Aliens are also known to particularly proliferate on the highly anthropogenic soils found on the banks of many urban rivers (Pyšek & Prach 1993).

A number of studies link the regularity of human activity within a location with the successful invasion of alien species. Pyle (1995) found that the frequency of aliens also increased with higher recreational use on the Potomac River floodplain and a study on tropical savannah and dry woodland reserves (Macdonald & Frame 1988) found that the number of alien species present on sites was positively correlated with visitor number. The high incidence of interruptions to the continuity of river corridors in urban situations because of road and rail bridges, and due to culverting, means it is difficult to predict confidently a higher rate of spread for aliens compared to within a non-urban situation, as might otherwise be expected. Differences in plant reproductive strategy may play a key role in determining which invasive species are most successful in different urban situations. However, it does seem likely that certain invasive species are better adapted to take advantage of the peculiarities of urban river corridors when compared to native hydrochorous species.

It is clear, from the research on both alien species and environmental degradation, that the criteria for designation of greenways as corridors may, in effect, be contradictory. To be specific, the requirement that they should be easily accessible and provide utility for the

public, (Birmingham City Council and Land Care Associates (LCA) Ltd 1997) does not, according to the evidence, sit comfortably with the target of nature conservation.

5.5 GREYWAYS

While greenways may be the focus of considerable attention for planners, a much greater density and diversity of less conspicuous linear features exist over most urban zones. Road verges, walls, rail track and pavements represent a dense interconnecting network of commonly highly disturbed and generally understudied communities. There is also a clear potential for interchange between the vegetation of these habitats and the early successional communities growing on the ephemeral derelict sites of the urban landscape. These habitats may contribute significantly to the diversity of a city. Thirty-six percent of native species classified as rare within the City of Nottingham were recorded upon derelict sites (Shepherd 1995). Aesthetically they may bear little comparison to the lush open green spaces of designated wildlife corridors, but in reality it is perhaps these open vegetation strips that truly exhibit the best example of habitat connectivity in cities.

It would however be wrong to dismiss the significance of these greyways for local biodiversity. In the UK, Sargeant's (1984) work clearly identified railways and their associated land as an important botanical resource. 1632 phanerograms (including aggregates, species, subspecies and varieties) were found to be present and over 200 of the vascular species recorded gave rise to one or more new 10km records. In cities, locally uncommon species may also be found on other linear habitats and clearly interconnections may ensure they remain in established populations and that they have the ability to spread to new locations and proliferate should conditions be favourable.

Dispersal and establishment in greyways

The evidence to support the spread of plant species dependent on animal vectors along these corridors is, as with greenways, predominantly by inference rather than direct evidence. Teagle (1972) noted the use of tree-lined boulevards by birds crossing the urban landscape in Birmingham and grey squirrels have also been found to use trees along roads to travel between wooded habitat patches (Gurnell 1987). Studies on small mammal (Vermuelen 1984)

and insect (Eversham & Telfer 1994) populations of roadside verges identify them as effective areas of habitat. Dispersal of plant seeds may be accelerated along the roadside corridor if foraging behaviour is more linear and directional within these habitat strips. However, while verges may provide important habitat, the actual road carriageway may be an effective barrier to dispersal for potential vectors (Mader 1984; Mader *et al.* 1990; Slater 1995).

A number of papers document the use of railway tracks by mammals. Page (1981) observed foxes using the edges of tracks leading in and out of London; Yalden (1980) found that wood mice reached into the city of Manchester along semi-natural vegetation adjacent to the track; and Chapman (1977) suggested that occasional sightings of Muntjac in urban areas could be explained by their use of these features. Animals are only likely to travel along these strips if they represent well established habitat in an otherwise unfavourable landscape. On new road verges along reclaimed polders in the Netherlands, zoochorous species of plant are consequently very rare and scattered in the initial stages of colonisation (Nip van Voort *et al.* 1979).

Due to the homogeneous nature of these strips, there is considerably greater evidence for anthropochorous dispersal by plant species along greyways than for greenways, as will be laid out below. The most successful migrants along these discrete interconnections will be well adapted to the highly modified environments they are confronted with. On road verges in cities tolerance of exhaust emissions and de-icing salt may be key (Gilbert 1989). De-icing coincides primarily with periods when rainfall exceeds evaporation and so is accompanied by almost continuous leaching (Thompson 1986). Higher levels of soil salinity due to this process have been recorded as far as 15 metres from the road edge (Jones 1981). A number of maritime plant species have been able to extend their range inland by spreading along road verges. The successful spread of these species has been well recorded on inland roads in the UK (Matthews & Davison 1976; Scott & Davison 1982, 1985; Roper 1996) demonstrating how some native species can gain an advantage from anthropogenic activity to increase their distribution. Dispersal is more rapid in the direction of traffic flow (Scott & Davison 1985) which points to the significant role motor vehicles play as a dispersal agent. Vehicles have been demonstrated as effective vectors of plant seeds by numerous researchers (Clifford 1959; Schmidt 1989; Lonsdale & Lane 1994; Hodkinson & Thompson 1991) by sampling in

mud on tyres or mud guards. Not surprisingly therefore the number of seeds transported per vehicle is much higher on non-surfaced rural roads (Hodkinson & Thompson 1991). However the higher density of traffic in the urban landscape means that the overall contribution may be equally or more significant. Small-seeded species are the major beneficiaries of motorised dispersal, being found in higher numbers (Hodkinson & Thompson 1991) and being transported further (Scott & Davison 1985) than large seeds.

Elevated levels of CO_2, NO_2 and SO_2 may be expected commonly recorded beside roads. Jones (1979). The zone of influence on the plant community is influenced by the density of traffic on the road (Angold 1997). Hunt *et al.* (1991) found that higher levels of CO_2 prevented late successional species achieving complete dominance over early successional species, so that a more open vegetation situation was likely to remain. Bazzaz & Garbutt (1988) similarly found that enhanced CO_2 encouraged the persistence of annual species.

Successful plants on UK railway tracks (Plates 5.2-5.4) must be able to tolerate and proliferate under highly xerothermic conditions associated with the ballast substrate and some degree of resistance to spraying will also be advantageous (Gilbert 1989). Resistance to spraying may incorporate a physiological tolerance to the herbicide or a life history pre-adapted to favour success, most obviously annuals with seed banks that can achieve rapid regeneration between spraying episodes. The decline in *Chaenorrhinum minus* and *Convolvulus arvensis* along UK railways is probably attributable to modern herbicide regimes. *C. minus* is still spreading successfully along US lines, where spraying is conducted later in the year, and *C. arvensis* remains abundant in the Isle of Wight where spray trains are not used (Sargeant 1984). As a consequence of these conditions, the communities found on the cinder track ballast are typically dominated by annual or short-lived perennials. However species which are able to persist may be accelerated in their spread and proliferation by vortices of wind created when trains pass (Arnold 1980).

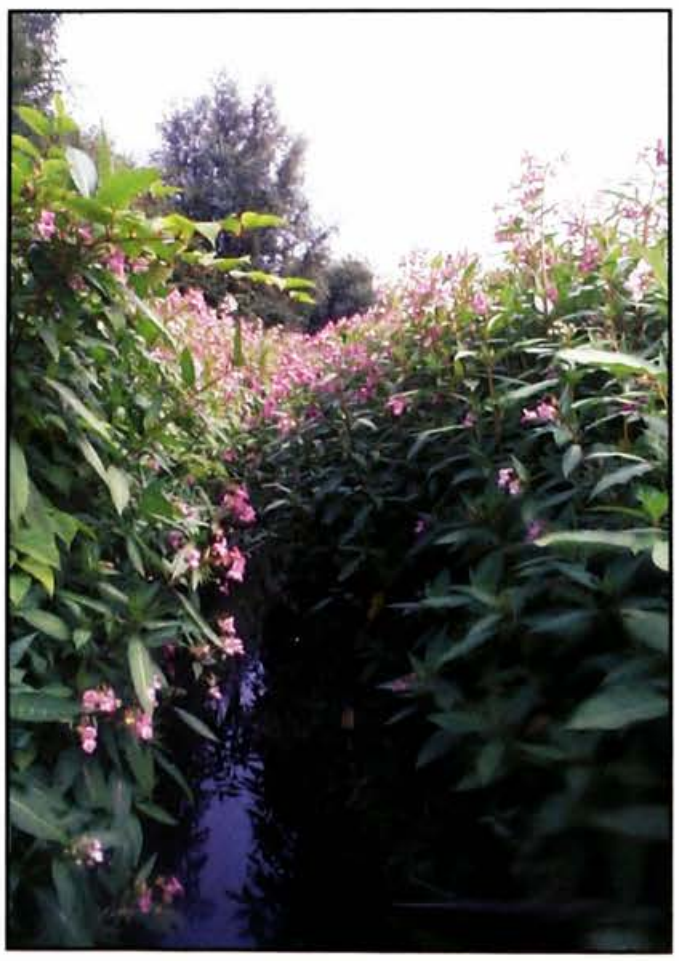
Retaining walls may also be common along either or both sides on urban railways and represent potential corridor opportunity for epiphytic species, most notably ferns of which a good proportion of the British species list may be found near railways (Dony 1953). The growing environment offered by different types of walls in urban areas is extremely variable both physically and chemically. Degree of exposure, aspect, seepage and presence of mortar

Plate 5.2: *A railway corridor in the West Midlands. While the rail track itself is clearly a homogeneous linear feature it can be seen that the embankment has a much patchier vegetative composition. Scrub, coarse grassland, tall herb and finer grassland communities are all visible*

Plate 5.3: Asplenium trichomanes, *an abundant species in western Britain, but a rare Local Biodiversity Action Plan species in the West Midlands, here seen growing on walls enclosing railway land in Smethwick. This species is a potential benefactor from urban greyways.*

Plate 5.4: Heracleum mantegazzanium, *another invasive alien commonly found along linear features in the West Midlands.*

Plate 5.5: Impatiens glandulifera, *an invasive alien species in the UK. Here shown growing with spectacular success beside the river Cole wildlife corridor in the West Midlands.*



are all significant for determining the species found. Some species, notably rusty back fern, *Asplenium ceterach*, are well adapted to deal with exposed warm and dry walls, while many mosses and liverworts are highly successful within shaded, damp locations.

For plants growing in cracks in pavements, the urban landscape is almost a continuous habitat, but the linear nature of these features is the reason for their inclusion here. Tolerance of trampling is the key to success on pavements. Low-growing rosette or caespitose species including *Taraxacum officinale* agg., *Poa annua*, and *Plantago major* are particularly successful under these conditions (Rodwell 2000).

Aliens on greyways

The spread of alien species correlates closely with human transportation and civilisations, so introductions can be traced back to the first significant movements of humans. Salisbury (1964) notes that the network of roads built during the Roman occupation must have been important pathways for the dispersal of weeds and aliens introduced from the continent. *Lythrum salicaria*, an introduction from Europe, can be a significant urban pest in North America (Hemming 1997). In New York state, its spread has been closely correlated with the Thruway (Wilcox 1989) with additional infrastructure features such as ditches and culverts aiding its invasion into surrounding wetlands. Numerous studies have also demonstrated that roads can be a direct threat to protected natural areas by providing the major route of invasion (Amor & Stevens 1976; Cowie & Werner 1993; Lonsdale & Lane 1994).

The use of railways as a means of dispersal and proliferation by alien plants is widely covered in the literature to date. In the UK, the spread of the alien *Senecio squalidus* from Oxford along railway lines has been well documented (Kent 1960, 1964), while Sargeant's (1984) comprehensive survey of railway vegetation found a number of alien species that are spreading (e.g. *Bunias orientalis*, *Barbarea intermedia*, *Epilobium brunnescens*), although whether this is accelerated by rail track has not been established.

There are a number of other alien species for which both roads and railways have been observed as a significant route for spread. The wool alien *Senecio inaequidens* has spread considerable distance along Netherlands transport links from its original escape despite most of the anemochorus seeds remaining near the parent (Ernst 1998). The summer annual *Erucastrum gallicum*, a crucifer native to Europe, dispersed rapidly across the US and Canada

along both railroads and roads after introduction in the early 1900s (Warwick & Wall 1998). One invasive species which frequents these linear habitats in the UK is giant hogweed, *Heracleum mantegazzanium,* which is frequent on railway embankments and verges where anthropogenic activity has altered the nutrient balance of the soil (Otte & Franke 1998).

Perhaps the most persuasive feature of the corridor function of pavements and trampled verges is the dispersal of pineapple weed, *Matricaria discoidea*, a species first recorded in the UK in 1871 but now abundant throughout. This species dependence on high intensity trampling that occurs on pavements is notable; where the impacts of humans are lower, it is replaced in the flora by the related native *Tripleurospermum inodorum.* Walls also can be an important habitat for aliens. Ivy leaved toadflax (*Cymbalaria muralis*) is another non-native species which is now familiar in the UK, predominantly found sprawling over walls or rockfaces. It appears likely that the density, interconnected distribution and linear nature of artificial habitats have played a significant role in shaping the current flora of the UK and will presumably continue to offer opportunity for future invaders.

5.6 Discussion

The evidence for native dispersal along greenways is generally poor. This is perhaps not surprising, as the typical species of the more stable woodland, scrub and grassland habitat types found within greenways are far less mobile than the early successional species found in greyways. They are species which, in general, have evolved over generations in continuous habitats where they have had no need to be mobile. Some protagonists of urban wildlife corridors could argue that the contemporary measurement of dispersal by plant species along corridors has not been conducted long enough to establish the facts. Yet Bastin & Thomas's (1995) study of woodland fragments in Birmingham found that proximity to historical greenways, since designated as wildlife corridors, had no impact on species diversity. They concluded that these 'narrow ribbons of habitat throughout urban areas', were 'probably ineffective' as a conservation measure due to the significance of edge effects and the slow dispersal rates of most species of conservation interest.

The evidence that invasive alien plants, particularly riparian species, could however be the major benefactors from increased connectivity of greenways suggests a possible scenario for

conservation that is somewhat worse than 'ineffective'. The high density of residential and public gardens within cities and the large amount of material transported into them means that in the modern age they are the predominant source of new invasions by foreign plants. Because of their natural or semi-natural components, greenways provide routes for effective alien dispersers from the city into the surrounding countryside, thereby potentially providing a route of spread to the few larger areas of pristine habitat that remain. The spread in recent times of *Fallopia japonica* and *Impatiens glandulifera* in the UK into sensitive riparian habitats is an indication of the dangers such an eventuality may hold.

It is evident that the potential for transfer of diaspores along the discrete greyways far exceeds that of most designated urban wildlife corridors. A higher degree of connectivity provided by their sheer density and homogeneity, and the faster mobility of the species found in these grey habitats contribute to this. Plants found along greyways are superbly pre-adapted to manipulate the intensity of human activity, and the effects of pedestrian, private or public transportation. The vast majority of species that are apparently benefiting are either widespread and abundant native species or alien species well-adapted to the highly modified urban environment but normally unlikely to become established in natural or semi-natural habitats. A few locally uncommon species finding greyway conditions analogous to their natural habitat may also utilise these linear features as refuge habitat and possibly dispersal pathways. A small percentage of the numerous alien species typical of these habitats do have the potential to become significant pests, although only *Heracleum mantegazzanium,* which has warranted the implementation of an eradication programme by the Environment Agency, is of much concern in the UK. These are dispersal corridors which are generally neither recognised by planning policy, nor in need of any protection, as their continued existence is entirely compatible with a changing urban landscape.

The time scale of most ecological research generally precludes contemporary measurement of dispersal by plant species along corridors and consequently historical records are often the most convenient means to demonstrate the spread of species. Where the actual rate of movement has been measured actively, the research generally focuses on fast-dispersing species which are invariably common and also spreading within the wider study area, and studies on scarce or even local plant species are rare. Consequently, studies examining the different extent of dispersal potentials of the individual species within assemblages or

communities are absent from the literature. If historical records are reliable and can be accurately related to equivalent modern data, then adopting the metapopulation approach to analysis can be more illuminating than simple measurement of spread or decline of populations. Comparison of extinction and colonisation of species within a group of patches has been found to be a good tool to determine the negative impacts of isolation within fragmented systems (Ouborg 1993; Fischer & Stocklin 1997). There is clearly potential to use this approach to consider whether connectivity, as provided by corridors, can mitigate these impacts. In the absence of historical data, the use of genetic markers could also be productive in identifying metapopulation structure (Milligan *et al.* 1994) and corridor effects. While research to support widespread dispersal along corridors by native species is rare, the evidence for spread of aliens along corridors is much wider and apparently more compelling. However, it should be remembered that these species are often, by their nature, more conspicuous to the native botanist and that their mechanisms and routes of dispersal have been considered of great significance and concern for some time. Consequently, research on changing distributions of these species has a longer and more extensive history. In contrast research on corridors as a means of maintaining native biodiversity *in situ* is comparatively recent, and perhaps it is therefore not surprising that there is limited field evidence at this time to support corridors for this function.

Currently, it is still only possible to speculate on the potential for corridors to provide connectivity between isolated populations of threatened species. The suggestion from much of the literature proposing the use of corridors and greenways within cities is that the 'precautionary principle' should be installed into our planning - expend resources on preserving and enhancing corridors now, while waiting to see whether further research justifies the outlay. However, the evidence currently available strongly suggests that the corridor concept when applied by planners in British cities is i) not relevant to urban linear green spaces, ii) not likely to provide any significant benefit to uncommon native plant species, and iii) may have the potential to hasten the proliferation of invasive alien species into sensitive surrounding habitats. It follows that each urban green site should be evaluated independently on the basis of the conservation importance of its component species or communities, and sites on greenways should not automatically be given priority over isolated sites.

CHAPTER SIX

THE RELATIVE IMPORTANCE OF ABIOTIC AND LAND USE FACTORS IN EXPLAINING THE VARIATION AND DIVERSITY OF VEGETATION ON DERELICT URBAN LAND

SUMMARY

The urban landscape is a highly complex mosaic, incorporating various human land uses and fragmented patches of remnant natural and semi-natural habitats, as well as many regenerative secondary communities. Theoretical and empirical research has shown that, within fragmented landscapes, site size and isolation may be influential in determining which species are present. Sites that are more connected to each other spatially might be expected to share more species in common, and sites that are particularly isolated may be expected to be less species-rich. These effects are recognised to occur at different scales for different taxa, and the response to isolation may depend on each species' own life history and dispersal characteristics. Considerable attention has also been paid to the potential fof linear habitats to mitigate the impacts of isolation.

This study examines the species assemblages on 50 derelict land sites across the West Midlands conurbation. Floristic affinity, measures of species richness and the representation of plant traits on these sites are examined in relation to the site's location within the urban landscape and proximity to similar habitat, railway track and river corridors. Site age, substrate and other indirect environment measures were also incorporated into the models used. A multiple regression model developed for similarity or distance matrices, which uses permutation testing of the regression parameters, was used to determine the best predictors of floristic affinity. Partial multiple linear regression was utilised to determine the best predictors for species richness measures and the representation of plant traits. Finally, the presence and absence of individual species was tested against various measures of site connectivity of sites using t-tests.

Environmental factors and site age were shown to account for most of the variation in the site assemblages that could be explained by the models. The landscape factors tested had only a minor impact on the floristic similarity of sites and no effect on species richness. The representation of traits at different sites did suggest a possible link with proximate dispersal routes but did not directly link dispersal capability with isolation. The individual species showing affinity to sites close to linear habitat 'corridors' were typically also those that regularly grow along these corridors. However, none of these species were rare in the urban zone and evidence does not suggest that they would otherwise struggle to disperse across the urban landscape. The implications of these results are discussed.

KEYWORDS: *Landscape ecology, Urban habitat, Corridors, Seed dispersal.*

6.1 INTRODUCTION

The prevailing environmental conditions of a location have long been considered the most significant determinant of the component plant species and their relative abundance in the vegetation (Curtis & McIntosh 1951; Bray & Curtis 1957). On successional seres, the time elapsed since the last major denudation or wide-scale disturbance may also be expected to be an important factor influencing both species richness and composition (e.g. Crowe 1979; Wilson & Tilman 2002) and changes in the vegetation over time can be linked closely to simultaneous changes in the site's environment (Crocker & Major 1955; Olson 1958). However, it is recognised that, under some circumstances, other factors may also be important. In particular, disturbance events, including fire, storms and disease, have been found to loosen the link between the environment and the vegetation found growing on sites (Wilson & Tilman 2002; Collins, Wien & Philippi 2001). In addition, theoretical and empirical research has shown that within fragmented landscapes, of which urban zones are a good example, site **size and isolation** may also be influential in determining which species are present. As discussed below, most recent work in this field has focused on the impact that fragmentation and isolation has on the diversity of species found, on identifying how different taxa respond to isolation, and on the extent to which isolation effects may be mitigated by better landscape management.

The impacts of fragmentation and isolation on plant communities

The study of habitat fragmentation in the last 35 years has been conducted largely within the framework of two key theoretical developments: the theories of island biogeography (MacArthur & Wilson 1967) and metapopulation dynamics (Levins 1969). The equilibrium theory of island biogeography is primarily concerned with the influences of island size and isolation on species composition. In particular, this theory postulates that, *other factors remaining equal,* the size of an oceanic island and its distance from a continental source of colonising species would determine the number of species present, since more distant islands would have lower species immigration rates, while smaller islands would have higher species extinction rates. This theory has been supported by empirical evidence in several studies of the flora of oceanic islands (Harvey, 1994; Moody 2000; Yeakley & Wieshampel 2000).

Metapopulation theory was originally conceived to describe the population dynamics of species occurring in naturally patchy habitats such as mountain tops (Brown 1971), where spatially separated populations undergo periodic colonisation and extinction. In contrast to island biogeography theory, metapopulation theory is primarily concerned with connectivity and interchange between spatially distributed con-specific populations (e.g. Hanksi *et al.* 1995), and mathematically predicting the probabilities of metapopulation population persistence where colonisation and extinction parameters are known. This model has been successfully applied to predicting the persistence of animal species which occur in terrestrial habitat fragments, such as red squirrels (Opdam *et al.* 1992), butterflies (Thomas, Thomas & Warren 1992) and birds (Villard, Freemark & Merriam 1989).

The direct application of these theories to the understanding of plant species distributions in terrestrial, and particularly urban, landscapes is problematic for a number of reasons (Forman 1995, Meffe & Carroll 1994). Firstly, urban landscapes are a highly heterogeneous mosaic, with many, particularly generalist, species able to survive at least at low rates in the matrix between patches. The diverse habitats within the mosaic, adjacent landscape elements and indeed the seed bank of the site itself, are all potential sources from which dispersal to the site can occur. Therefore, the location of and distance between source pools, a primary characteristic of both island biogeography and metapopulation theory, is difficult to measure meaningfully. Arguably, isolation will also only be a major problem for a minority of species

(Forman 1995).

Secondly, an early criticism of island biogeography theory was that since most patches contain internal habitat diversity, the predicted 'area-*per-se'* effects of patch size on species richness are often difficult to demonstrate (Simberloff 1976; Simberloff & Gotelli 1984). For example, as discussed above, disturbance events within the patch, which are generally frequent occurrences in cities, play a major role in determining species richness. This internal habitat diversity and additionally, inter-patch differences in environment, origins and patch persistence, violate the assumption of island biogeography theory that "other factors remain equal".

Other problematic issues in city landscapes include the fact that few, if any, urban habitats are likely to be 'in equilibrium', since the landscape by definition has been recently modified. In addition, the typically small size of urban fragments mean that the edge-effect and interior-to-edge ratio, also present on islands, are potentially much more of an important factor to explain some species-area patterns than immigration and extinction rates (Forman 1995).

In this context, it is perhaps unsurprising that the numerous studies conducted to investigate the effects of patch size and isolation on plant assemblages and species in terrestrial fragmented systems, have produced conflicting results. Patch isolation was found to be relatively unimportant in explaining plant species richness in fragmented mires (Ouborg 1993) or farm woodlands (Usher, Brown & Bedford 1992), while it was shown to more significant in other studies in woodland (Galli *et al.* 1976; Bastin 1997; Jacquemyn, Butaye & Hermy 2001), chaparral (Soule *et al.* 1992), and in several studies of the metapopulations of individual plant species (Giles & Goudet 1997; Eriksson & Kiviniemi 1999; Husband & Barrett 1998; Senneville *et al.* 2001). Habitat specialists have been shown to be more vulnerable to extinction than habitat generalists for a variety of habitats (Rathcke & Jules 1993; Fischer & Stocklin 1997; Grashof-Bokdam 1997). Generalist species can be expected to be at a strategic advantage over specialists in fragmented populations because of their wider ability to exist within the matrix of surrounding habitats, which by definition is inhospitable to a greater degree for a specialist plant of isolated habitat. This means that they have a wider source base and therefore have individuals closer situated to colonise an unoccupied site. As time of arrival may be crucial to successful establishment, this is a

particularly important advantage, even where the isolation of the habitat specialist may not prevent it dispersing seed given time and favourable circumstance.

How different taxa respond to isolation: the importance of seed characteristics

A potentially effective approach to comparing the patterns of site occupancy exhibited by plant species in patchy environments within a spatially heterogeneous landscape is to consider their life history traits (Amarasekere & Possingham 2001). Colonisation ability in particular might be expected to be particularly important in determining which species are present in habitats which have undergone significant fragmentation (Eriksson & Eriksson 2000). As was demonstrated in Chapter 3, the majority of species found in the early successional habitats of derelict land are well-adapted to dispersal and colonisation. However, differences may still be expected between sites based on how isolated they are particularly on older sites.

The vast majority of seeds of most plant species fall within a few metres of the parent plant (Verkaar 1990). However, amongst species with better dispersal capabilities there is greater potential for longer distance dispersal by a small percentage of seed from an individual plant. These longer distance events will be determined by a combination of these seed characteristics, the prevailing weather (particularly wind and rain), and chance events such as the intervention of humans.

Seed weight is perhaps the single most important plant characteristic in determining dispersal potential. The most successful terrestrial long-distance dispersing flowering plant taxa are those with minute seeds such as *Juncus* spp. and members of the Orchidaceae. Non-flowering plants, such as ferns and mosses, typically also have spores that can drift high into the air currents and be carried over vast distances (e.g. Delgadillo 1993; Skotnicki, Ninham & Selkirk 1999). There is also evidence that having small seeds may be a particular advantage for dispersal across cities. Studies on the seeds found on cars and bicycle tyres have shown that the majority are small-seeded species (Hodkinson & Thompson 1997). Another adaptation that increases the distance that a seed can float in the air is the reduction in terminal velocity by having increased air resistance due to appendages such as plumes (e.g. members of the Asteraceae).

An additional disadvantage of the large-seeded strategy is the trade-off incurred with seed production. However, if large-seeded species are harnessed with an edible seed coat to attract

animals and especially birds, then dispersal may be much further. Alternatively, coats may have modified hooks or barbs, which grasp to the fur of animals and can be carried for some distance before working loose. However, the poor numbers of zoochorous species on early successional habitats (Hodgson & Grime 1988; Prach, Pyšek & Šmilauer 1999) suggests they are typically less effective at arriving quickly at sites. This may be partly because for birds there are no suitable perching locations in many open habitats (McDonnell & Stiles 1983) and because, for smaller mammals, these locations are too exposed to predators. However, even in old stands of secondary woodland, it is mostly zoochorous plant species that are under represented, compared to ancient woodlands (Peterken 1974).

The ability to persist as a dormant seed for long periods may also be beneficial within patchy ephemeral habitats such as derelict land. This benefit may be related directly to the nature of derelict land itself, particular the haphazard disturbances which enable a pioneer species with a persistent seed bank to continually re-establish themselves after being suppressed by competitive later successional vegetation (see Chapter 4). However, particularly in urban areas seed persistence can also be viewed as an advantageous strategy for colonising a site at the outset of succession. Species with a long-term persistent seed bank may be present in underlying soil which is exposed during demolition and bulldozing of sites. Species which form large seed banks are also at an advantage at the beginning of succession on refuse or toxic sites which are capped with clay and topsoil if they are contained within the in-filled material. Seed bank-forming species are also more likely to be represented in dumped garden soil and thrown out pots (Hodkinson & Thompson 1997).

Mitigating the impacts of isolation

The mechanisms proposed to counter the decline in species richness that might occur in a fragmented system can be direct or indirect. The monitoring and management of isolated sites as nature reserves to conserve sensitive species can be viewed as an intensive direct response to habitat fragmentation. The practical approaches to and effectiveness of direct conservation management in the UK have been much reviewed (e.g. Ratcliffe 1977; Spellerberg, Goldsmith & Warren 1991) and formed the basis for most conservation policy strategists in the last half of the 20th century. This approach could also entail the replacement of lost populations of species in isolated sites via the translocation of individuals or sowing of seed

from plants at sites where populations remain stable.

In the last twenty years, much theoretical consideration has been given to the desirable concept that better landscape planning could enhance the potential for natural dispersal of propagules between sites and reduce the need for, or at least the extent of, direct intervention. A better spatial understanding of habitat patches may also enable sensitive sites and taxa to be highlighted and targeted for particular need of direct intervention (Forman & Godron 1984). This understanding could also be beneficial when considering sites for habitat recreation schemes, the potential for which is now high in some agricultural areas of the UK. Amongst options which may successful mitigate against fragmentation, considerable focus has been directed to the concept of ensuring the retention, or reimplementation, of linear habitat strips between isolated patches (Diamond 1975; Wittig & Schreiber 1983; Noss 1983; Merriam 1984; Forman & Godron 1984, 1986).

These linear features have been termed 'wildlife corridors' and many attributes have been assigned to them in the scientific literature. Corridors certainly have an aesthetic value playing an important role for human recreation (Adams & Leedy 1987; Burgess *et al.* 1988; Barker 1995; Forman 1995). They may also offer important and distinct habitats in their own right (Simberloff *et al.* 1992; Forman 1995; Hess & Fischer 2001) and consequently contribute significantly to the overall habitat area of a region (Andrews 1993). However it is the extent to which corridors may act as 'biogeographic land bridges' which is still very much open to debate. Numerous authors (e.g. Diamond 1975; Wittig & Schreiber 1983; Simberloff *et al.* 1992; Forman & Godron 1986; Forman 1995) have suggested that by aiding the dispersal of species and facilitating the intermixing of propagules between remnant isolated patches linear features can indeed play that role. However, despite considerable active research, this function of corridors has only been clearly demonstrated for a few taxa. The scientific evidence for and against the utility of corridors within the urban zone for the dispersal of plants has been more thoroughly reviewed within Chapter 5.

Derelict habitats and spatial ecology

Derelict sites are typically spatially discrete, having obvious boundaries. They have been described as analogous to islands, being isolated in a sea of asphalt and concrete (Pickett 1976). They are also often linked to linear urban features, particularly industrial transport

links such as railways and canals. In addition, the date at which derelict sites became derelict and were graded or were denuded can generally be traced. Because of this, derelict plots provide useful habitats for landscape related ecological study, as the majority of species, (other than those in the seed bank of the original site or transported soil) must necessarily have colonised the site over a known time period.

Aims of the study

In this study the relationship between the plant species found on urban derelict land patches and environment, site age and spatial factors is examined. The aim is to elucidate the extent to which landscape measures are able to predict the flora found in these habitats and consequently determine the significance of isolation and proposed mitigating solutions such as corridors.

The approach to investigating these issues was to:

(i) examine to the extent to which floristic similarity between pairs of sites can be predicted by their abiotic circumstances and spatial location;

(ii) determine which of the factors that may influence a species assemblage best predict the diversity found at a site, both in terms of species richness, density and other measures of floristic significance;

(iii) investigate how the representation of key plant characteristics or traits respond to different measures of site isolation; and

(iv) look directly at certain measures of site isolation to identify which species show a positive (or negative) association with greater connectivity.

6.2 METHODS

Floristic data

Field surveys took place at 50 derelict land sites between June-September in 1998 and 1999. Sites were located across the West Midlands conurbation and were selected from database information provided by local councils and on the basis of observation while driving around the region (see Appendix I for site list). Site selection, discussed in Chapter 2, attempted to

represent the diversity of different types and ages of derelict land, while ensuring a relatively even geographical spread of sites so that inner city, suburban and urban fringe habitats were all sampled. Sites were located in areas with both a high- and low-density of surrounding derelict land (Figure 6.1), and 25 of the 50 sites were located within 500 metres of a railway line. Walking transects were undertaken across the whole site at 5 metre intervals to draw up a complete list of higher plant species present (Appendix IX). Nomenclature is according to Stace (1997).

These data were used to calculate a number of floristic variables as listed in Table 6.1a. Total site species richness (TOTRICH) and mean quad species richness (QUADRICH) were calculated. The number of native species at the site (NATIVERICH) was established using data from forthcoming Atlas of the British Flora (Preston, Perman & Dines *in press*). Species distribution data from the wildlife database for the Black Country and Birmingham (Ecorecord) was used to identify the native species which were uncommon or rare within the region (found in fewer than 6 km sq.). The number of such species (RARERICH) and their contribution to total richness (%RARE) was then found for each site. Finally a specialism score for each site (MEANSPEC) was calculated by taking the mean value for the species indices calculated by Thompson, Hodgson & Gaston (1998), for which increasing value indicates higher habitat fidelity.

The representation of some key plant characteristics amongst the species assemblages was determined from a number of existing sources. Seed weights were taken from Grime *et al.* (1981), seed longevity is according to Thompson, Bakker & Bekker (1997), ruderality scores from Grime Oikos 1997 paper, and presence of zoochorous adaptations are from Grime, Hodgson & Hunt (1988). A site mean value was taken for seed weight, longevity, and ruderality for all the species present for which a measure was available (SEEDWEIGHT, SEEDLONGEV, RUDERALITY). The zoochory score was calculated as the percentage of species present at the site known to have zoochorous adaptations (%ZOOCHOR).

Environmental data

Obtaining direct measurements for a suite of environmental factors at such a large number of sites was prohibitive due to time and resource constraints. Additionally, direct sampling of soil characteristics would have been problematic due to the high heterogeneity typical of

many derelict sites. Instead, indirect environmental indicator values for nitrate, salinity, pH, soil moisture and light were used were based on Ellenberg's *Zeigerwerte* (Ellenberg, 1979, 1988; Ellenberg *et al.* 1991), recalculated to explain ecological ranges of species for Britain and Ireland (Hill *et al.* 2000). These values have been used to infer environmental conditions in continental European research (Prach, Pyšek, and Šmilauer 1999). The mean Ellenberg values for species in a series of 1m x 1m quadrats taken at each site were calculated (for further details see Chapter 2 Methods). The use of the quadrat data rather than the species list to obtain these measures meant that the values obtained were largely independent of the total site assemblage data used in the analyses. The mean value at each quadrat within a site was then used to derive the site mean values (listed in Table 6.1b).

The predominant substrate type or types were also recorded for each site. This was based on information derived from numerous sampling cores taken at each site during the seed bank study (Chapter 4). Substrates categorised were urban topsoil (URBTOP), agricultural topsoil (AGRICTOP), clay infill (CLAYINFILL), brick rubble (BRICK), ballast (BALLAST), sand (SAND) or broken concrete and tarmac (CONCTAR).

Site age since dereliction is also utilised here (AGE) and was obtained from information within the derelict land database (1998 version) of the Joint Development Team for the West Midlands (JDT); aerial photography for the region (CityView 1995); dated old Ordnance Survey maps and questionnaire replies from local residents (see Chapter 3 for methods and Appendix IV for site age data). Sites ranged in age from 2 to *circa* 20 years since dereliction.

Measures of surrounding landscape

The land-use/land cover of the area surrounding each site was quantified within buffers at three distances from the site boundary: 100m, 1km and 5km. The land cover within the 100m radius from the boundary was mapped in the field between August-October 1999. Each map was digitised, and incorporated into an ArcView GIS system (Environmental Systems Research Institute, Copyright 1992-1999, Version 3.2). This program was then used to extract the percentage cover of urban and suburban land and the percentage of 'similar' (derelict) land within the 100m buffer (URB100, DER100). ArcView was also used to provide accurate measures of site area (LOGSIZE). Appendix Xa provides full data from ground mapping.

At the wider scales, data from the JDT (1998) register of derelict land in the West Midlands

Figure 6.1: *The 50 derelict survey sites, and the density of derelict land within the West Midlands conurbation.*

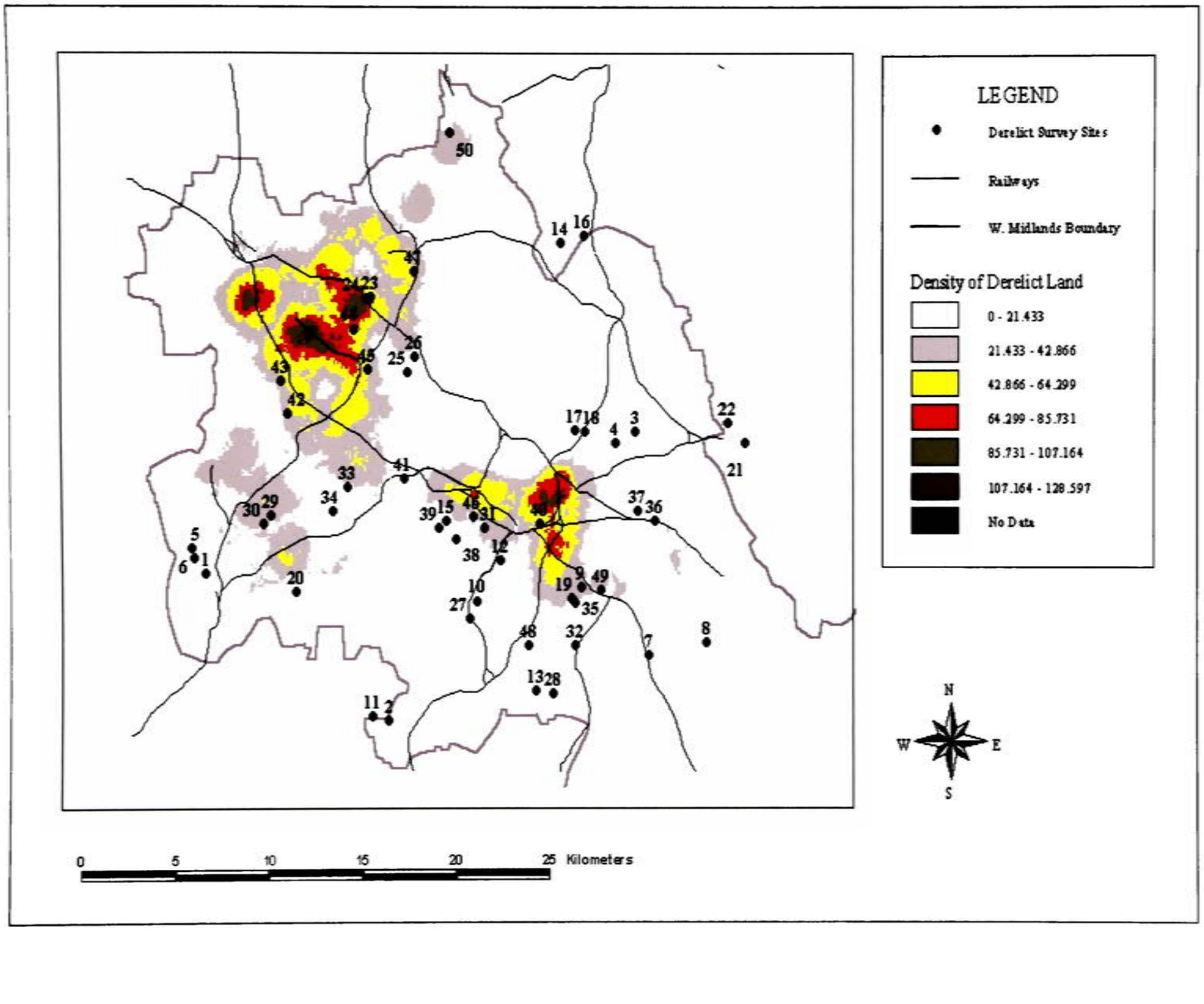


Table 6.1: *Table of floristic, environmental and landscape variables used in analyses*

(a) Floristic data (Y)

VARIABLE	Description	Source	Analysis where used: Mantel	Linear	Species
FLORA	Sørensen's coefficient of similarity between pairs of sites based on presence/absence of species	Field survey, site species lists K. Austin 1998-9	✓		
TOTRICH	Total number of species at site	"		✓	
NATIVERICH	Total number of native species	"		✓	
QUADRICH	Mean quad species richness	"		✓	
MEANSPEC	Mean specialism of species at site	"		✓	
RARERICH	Number of regionally rare species at site	"		✓	
%RARE	% of total species list that is regionally rare	"		✓	
SEEDWEIGHT	Mean seed weight of all species present	"		✓	
SEEDLONGEV	Mean seed longevity score of all species	"		✓	
RUDERALITY	Mean ruderality score of all species	"		✓	
%ZOOCHOR	% of total species list that is zoochorous	"		✓	
Species names	Presence/absence data for individual species	"			✓

(b) Environment data (X)

VARIABLE	Description	Source	Analysis where used: Mantel	Linear	Species
REACTION	Site average of mean quadrat pH (R) Ellenberg values, range 1-9	Field survey quadrat data, K. Austin 1998-9		✓	
LIGHT	Site average of mean quadrat light (L) Ellenberg values, range 1-9	"		✓	
MOISTURE	Site average of mean quadrat moisture (M) Ellenberg values, range 1-9	"		✓	
NITROGEN	Site average of mean quadrat fertility (N) Ellenberg values, range 1-9	"		✓	
SALINITY	Site average of mean quadrat salinity (S) Ellenberg values, range 0-3	"		✓	
ENVT	Gower's coefficient of similarity between pairs of sites based on mean site Ellenberg values R, L, M, N & S described above.	"	✓		
URBTOP AGRICTOP CLAYINFILL BRICK BALLAST SAND CONCTAR	Whether substrate is (1) or isn't (0) urban topsoil, agricultural topsoil, clay infill, brick rubble, ballast, sand or broken concrete/tarmac	Soil core field surveys, K. Austin 1998-99		✓	

***Table 6.1(b)** continued...*

SUBS	Similarity in substrate (URBTOP, AGRICTOP, CLAYINFILL, BRICK, BALLAST, SAND, CONCTAR) between pairs of sites calculated using Sørensen's binary coefficient	Soil core field surveys, K. Austin 1998-99	✓		
AGE	Age of site (years) since dereliction or major denudation event. For Mantel analysis, similarity in age between pairs of sites calculated by Gower's coefficient.	Questionnaires, JDT data, aerial photography, OS maps.	✓	✓	

(c) Landscape data (W)

			Analysis where used		
VARIABLE	Description	Source	Mantel	Linear	Species
LOGSIZE	Log of the site area. For Mantel analysis, similarity between pairs of sites calculated by Gower's coefficient.	OS maps	✓	✓	
GEODIST	Distance in kilometres between pairs of sites	"	✓		
URB100	Percentage of concrete cover within 100m buffer around site boundary	Ground mapping, K. Austin 1999		✓	
URB1000	Percentage of urban and suburban cover within 1km buffer around site boundary	LCMGB		✓	
URB5000	Percentage of urban and suburban cover within 5km buffer around site boundary	"		✓	
DISTEDGE	Shortest distance from site boundary to established edge of conurbation (km)	OS maps		✓	
URBCOV	Similarity between pairs of sites based on urban cover (URB100, URB1000, URB5000, DISTEDGE) calculated using Gower's coefficient.	Ground mapping (K. Austin. 1999) & LCMGB	✓		
DER100	Percentage of surrounding derelict cover within 100m buffer around site boundary	Ground mapping, K. Austin 1999		✓	
DER1000	Percentage of surrounding derelict habitat within 1km buffer around site boundary	JDT data		✓	
DER5000	Percentage of surrounding derelict habitat within 5km buffer around site boundary	"		✓	
DISTSIM	Shortest distance to nearest derelict patch	"		✓	✓
SIMHAB	Sørensen's binary coefficient of similarity based on distance to nearest derelict patch and thresholds of 100m, 500m & 750m.	JDT data	✓		
DISTRAIL	Distance to nearest railway (m). For Mantel analysis, similarity of DISTRAIL between pairs calculated using Sørensen's binary coefficient and thresholds of 100m, 300m & 750m.	OS maps	✓	✓	✓
DISTRIV	Distance to nearest river (m). For Mantel analysis, similarity of DISTRIV between pairs calculated using Sørensen's binary coefficient and thresholds of 100m, 300m & 750m.	OS maps	✓	✓	✓

was added to the ArcView GIS, and used to determine the percentage of derelict land within 1km and 5km buffers around the site boundaries (DER1000, DER5000). The shortest distance to a neighbouring derelict site was also determined (DISTSIM). Data from the Landcover Map of Great Britain (Fuller *et al.* 1994), derived from satellite data with 25m resolution, was used in ArcInfo to calculate the percentages of urban and suburban land cover within 1km and 5km buffers around the site boundaries (URB1000, URB5000). Appendix Xb-c summarises ArcInfo output and index calculations.

The proximity of each site to the nearest railway and nearest river was measured in metres using Ordinance Survey maps (DISTRAIL, DISTRIV). The OS maps were also used to establish the boundary of the conurbation, and the shortest distance between site and the urban edge was measured in kilometres (DISTEDGE).

A full list of the landscape variables used is given in Table 6.1c.

6.3 DATA ANALYSIS PROCEDURES

The four aims of the study outlined above required the use of several different statistical techniques as outlined in Figure 6.2. Each method of analysis, with the exception of the species distribution analysis, involved partitioning the variance in the response variable Y (whether it be floristic similarity, measures of diversity, or measures of trait representation) according to the set of explanatory environmental variables X and the set of explanatory landscape variables W. A large degree of autocorrelation between X and W, i.e. spatial structuring of environmental variables, is a major source of false correlations, which are not indicative of causal relationships (Legendre & Legendre 1998). If Y and W alone are examined, Type I errors may result, while if Y, X and W are examined simultaneously, the likelihood of these errors occurring is reduced (Legendre & Legendre 1998).

Different techniques of causal modelling are thoroughly discussed in Legendre & Legendre (1998). They include partial canonical analysis, univariate partial regression, logistic partial regression analysis and partial mantel analysis, and the choice of technique is determined largely by the form of the response and explanatory variables. The techniques employed in this study are discussed below.

(i) Modelling floristic similarity

Mantel tests and partial Mantel analyses was used to determine the best predictors of floristic similarity between sites. Ten distinct similarity/distance matrices were constructed, listed in Table 6.1. Sørensen's binary similarity coefficient (Sørensen 1948) was used to compute the similarity matrices of the flora (FLORA) and the site substrate (SUBS). This coefficient excludes double zeros, so that sites are considered similar only if a factor is present at both. The distance in kilometres between pairs of sites was calculated by trigonometry using the Northing and Easting of the NE corner of each site (GEODIST). The Gower similarity measure (Gower 1971) was utilised to provide matrices for environment site age (AGE) and log site area (LOGSIZE). This measure makes it possible to combine quantitative, semi-quantitative and qualitative values into a similarity measure and also incorporates normalisation of the data within its calculation (Legendre & Legendre 1983; Gower & Legendre 1986; Legendre & Legendre 1998).

Correlation between the independent variables collected for the analysis was shown to be high within the Ellenberg environmental variables, measures of urban land cover and measures of derelict land cover (Table 6.2a-c). Therefore, the Gower similarity measure was used to calculate a single matrix for environment (ENVT) based on all Ellenberg values, and a single matrix for surrounding urban land (URBCOV) based on all urban cover measures.

Sites that are more interconnected with linear features and/or each other may be expected to have more species in common for two reasons. Firstly, they may share in common some of the less mobile species; secondly, at a local scale, species may actually disperse between neighbouring sites. Conversely, it is not necessarily expected that sites that are more isolated from each other and/or from linear features will be more similar to each other. Isolated sites may be similar in terms of the *absence* of less mobile species, but they are less likely to have interchange of species at a local scale. Consequently, for connectivity matrices of distance to corridor (DISTRAIL, DISTRIV) and similar habitat (SIMHAB) a Sørensen's binary similarity coefficient was used, so that sites close to corridors or other derelict density areas were considered similar while those far from corridors or other derelict sites were not. Various threshold distances were applied in order to identify the scale at which any effect might be best observed. The threshold distances utilised from rail and river corridors were 100m, 300m

and 750m, while those used for distance to similar habitat were 100m, 500m and 750m.

Standardised Mantel (1967) tests were used to establish the coefficients of correlation between the resemblance matrices. This test has also been referred to as the Quadratic Assignment Procedure (Hubert & Schultz 1976). Both the Mantel t-test approximation from the Mantel statistic and a permutation test were used to calculate the significance of the coefficients. The permutation test repeatedly simulates realisations of the null hypothesis through permutations of lines and columns in the Y matrix and recomputes the Mantel statistic. If the null hypothesis is to be dismissed, then the original value of the Mantel statistic will be more extreme than most of the recomputed values (Hope 1968).

The best combination of predictors for floristic similarity were identified using a method of multiple regression developed for similarity or distance matrices which uses permutation testing of the regression parameters (see Legendre, Lapointe & Casgrain 1994). This was carried out within the Permute! software (version 3.4 alpha 9, Legendre, Lapointe & Casgrain 1994) on a Macintosh microcomputer. Permutations were carried out in the same manner as the Mantel (1967) permutational test calculations (above) and probability was computed according to Hope (1968). Forward selection and backwards elimination procedures were used (Figure 6.3), with selection or elimination dependent on whether the probability value is below or above a predetermined value (for our purposes; $p=0.10$) after Bonferroni correction (Cooper 1968, Miller 1977) for multiple testing. If, during the selection procedure, a tie occurs in the probability values based on the permutations test, then firstly the partial regression coefficient, and secondly the value of the increment in R^2 are used as selection criterion. For DISTRAIL, DISTRIV and SIMHAB, only the threshold distances which provided the highest coefficient in the previous Mantel procedure were included in the model.

(ii) Modelling floristic diversity

Measures of floristic diversity (TOTRICH, QUADRICH, NATIVERICH, RARERICH, %RARE and MEANSPEC, see Table 6.1a) were modelled using partial multiple linear regressions. A stepwise selection procedure was used to select the best environmental and landscape explanatory variables, using criteria of $p<0.05$ to include and $p>0.10$ to remove.

Figure 6.2: *Aims of study and methods of data analysis*

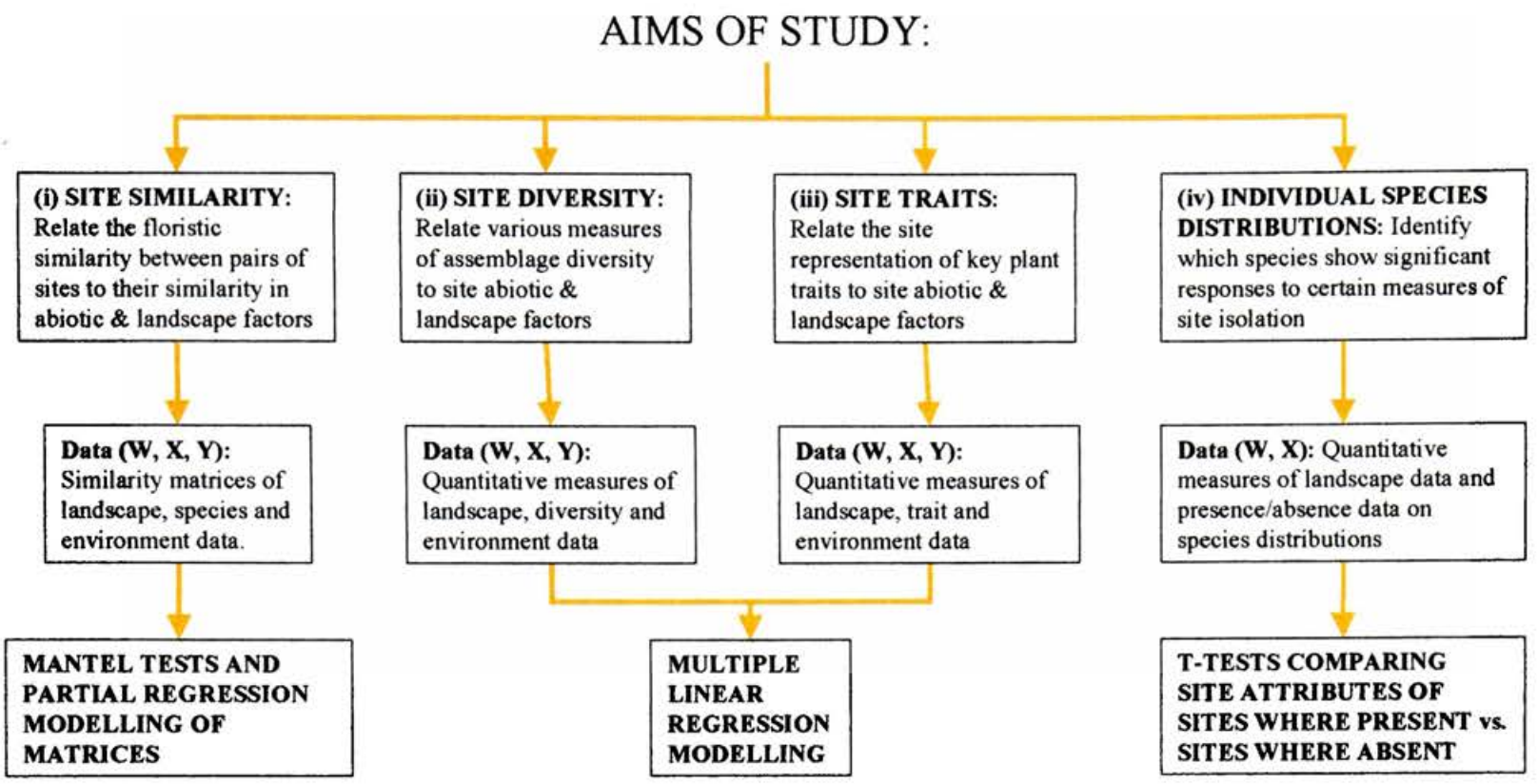


Figure 6.3: *Backward and forward methods for selecting a subset of predictors. Selection is based on permutational probabilities. From Legendre* et al. *(1994)*

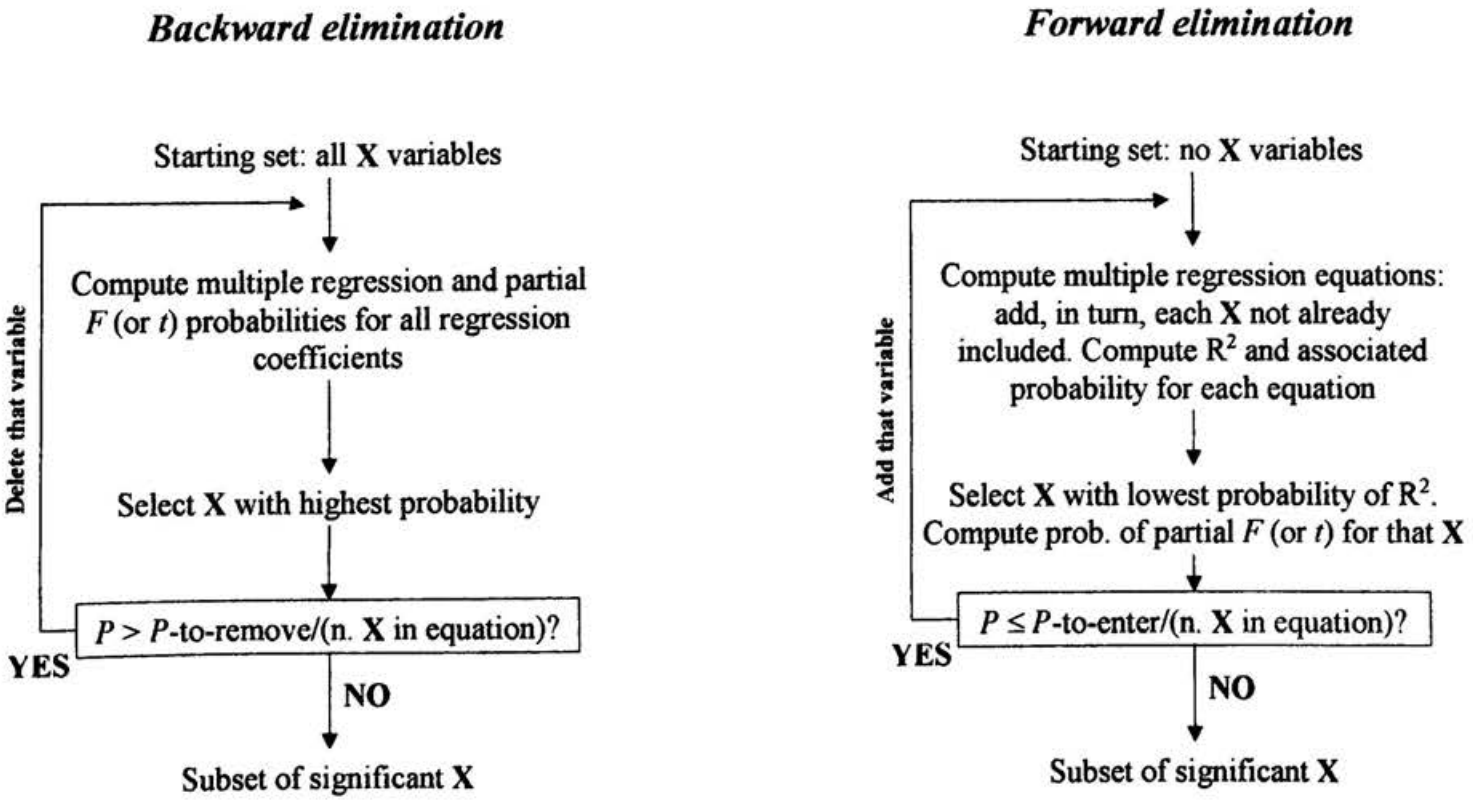


Table 6.2: *Spearman's rank correlation coefficients demonstrating similarity between different site variables a) Ellenberg values b) surrounding urban land cover c) surrounding derelict land cover. (* $p<0.05$; ** $p<0.01$; *** $p<0.005$; **** $p<0.001$).*

a) Ellenberg values

	LIGHT			
MOISTURE	*-0.676* ****	MOISTURE		
REACTION	*0.386* ***	*-0.286* *	REACTION	
NITROGEN	*-0.300* *	*0.371* ***	*0.357* ***	NITROGEN
SALINITY	*0.162* ns	*0.144* ns	*0.157* ns	0.322 *

b) Surrounding urban land cover

	DISTEDGE		
URB5000	*0.812* ****	URB5000	
URB1000	*0.580* ****	*0.714* ****	URB1000
URB100	*0.340* **	*0.346* **	*0.477* ****

c) Surrounding derelict land cover

	DISTSIM		
DER5000	*-0.676* ****	DER5000	
DER1000	*-0.799* ****	*0.647* ****	DER1000
DER100	*-0.114* ns	*-0.042* ns	*0.147* ns

(iii) Modelling trait representation

Measures of trait representation (SEEDWEIGHT, SEEDLONGEV, RUDERALITY and %ZOOCHOR, see Table 6.1a) were also modelled using partial multiple linear regressions. A stepwise selection procedure was used to select the best environmental and landscape explanatory variables, using criteria of $p<0.05$ to include and $p>0.10$ to remove.

(iv) Individual species distributions

Two-tailed T Tests were used to compare the mean connectivity of sites at which species were present with those where they were absent. This was performed using (i) distance from rail corridor (DISTRAIL); (ii) distance from river corridor (DISTRIV); and (iii) distance from nearest derelict site (DISTSIM). All species found at a frequency of 10% (5) or more sites were tested in this manner against all three variables.

6.4 RESULTS

(i) Floristic similarity

Correlation coefficients between the resemblance matrices as calculated by the standardised Mantel test and Bonferroni corrected significance levels are given in Table 6.3. Sites that were similar in their environment (ENVT, $p<0.001$) and / or substrate (SUBS, $p<0.001$) were found to be significantly more similar to each other in terms of their flora (FLORA). Sites of similar age (AGE, $p<0.05$) and with other derelict sites within 500m (SIMHAB, $p<0.01$) were also more closely akin in species composition.

Correlation between the other variable matrices was quite low. After the Bonferroni correction is applied ENVT & SUBS, and SUBS & AGE were significantly positively correlated with each other. There were however no significant correlations between the environment (X) variables and the landscape variables (W), indicating that there was little autocorrelation between W and , i.e. little spatial structuring in the environment variables.

The results of forward selection and backward elimination procedures used in the Permute! software are given in Table 6.4a & b. In each procedure, the best predictors of floristic similarity were shown to be the same four variables that were identified by the individual

Table 6.3: *Correlation table for resemblance matrices derived from standardised Mantel tests. Figures in italics represent Mantel* r *statistic. Figures in bold represent probability of conforming to the null hypothesis (H^0) tested by 9999 permutations. Significant correlations after Bonferroni corrections are applied are shown by asterisk. (* p<0.05; ** p<0.01; *** p<0.005; **** p<0.001). For connectivity measures only the threshold distances providing highest coefficients are shown.*

	FLORA (Y)	ENVT (X)	SUBS (X)	AGE (X)	SIZE (X)	DISTRAIL (W)	DISTRIV (W)	URBAN (W)	SIMHAB (W)
ENVT (X)	*0.4587* **0.0001 ******								
SUBS (X)	*0.1755* **0.0001 ******	*0.1281* **0.0021 ***							
AGE (X)	*0.1188* **0.0157 ***	*-0.0210* **0.3324**	*0.1498* **0.0004 *****						
LOGSIZE (X)	*-0.0259* **0.3616**	*0.0798* **0.1317**	*0.0041* **0.4285**	-0.0631 **0.0612**					
DISTRAIL (W) (300m threshold)	*0.1838* **0.0133**	*0.0877* **0.1528**	*0.0146* **0.3265**	*-0.0368* **0.2508**	*-0.0557* **0.2066**				
DISTRIV (W) (300m threshold)	*0.0460* **0.2850**	*0.0001* **0.5095**	*0.0440* **0.1355**	*-0.0541* **0.1326**	*0.0001* **0.4780**	*0.0291* **0.4281**			
URBCOV (W)	*0.1127* **0.0588**	*0.0146* **0.3881**	*0.0839* **0.0205**	*0.0642* **0.0956**	*-0.0420* **0.2446**	*-0.0140* **0.4335**	*-0.0140* **0.4335**		
SIMHAB (W) (500m threshold)	*0.2841* **0.0011 ****	*0.0700* **0.2196**	*0.0780* **0.0382**	*-0.0321* **0.2915**	*-0.0071* **0.4703**	*0.0528* **0.3432**	*-0.0874* **0.1975**	*0.0720* **0.1640**	
GEODIST (W)	*-0.1184* **0.0337**	*-0.0649* **0.1675**	-0.0375 **0.1494**	*-0.0829* **0.0352 ***	*-0.0037* **0.4541**	*-0.0046* **0.4809**	*0.0001* **0.0512**	*-0.1784* **0.0014 ***	*-0.0361* **0.2890**

Table 6.4a *Backward elimination procedure for selecting an optimal subset of explanatory matrix variables for derelict site species assemblages. The model parameters (Std. b) are standard partial regression coefficients. All probabilities (P) are one-tailed, computed after 9999 random triple permutations of the dependent matrix variable; minimum probability for each model is therefore 1/(9999+1) = 0.0001; At each step the variable chosen for elimination is marked with a cross (†); the variable with the largest probability is eliminated if its probability is larger than the Bonferroni-corrected significance level p = 0.10/{number of variables in the model at the given step}.*

MATRIX VARIABLE	STEP 1		STEP 2		STEP 3		STEP 4		STEP 5		STEP 6	
	Std. *b*	P	Std. *b*	P	Std. *b*	P	Std. *b*	P	Std. *b*	P	Std. *b*	P
ENVT	0.416	<0.001	0.413	<0.001	0.413	<0.001	0.413	<0.001	0.416	<0.001	0.428	<0.001
SIMHAB (500m)	0.245	0.003	0.245	0.001	0.249	0.001	0.243	0.001	0.245	0.001	0.251	0.001
AGE	0.103	0.010	0.105	0.009	0.108	0.008	0.110	0.010	0.110	0.009	0.104	0.013
SUBS	0.074	0.017	0.074	0.010	0.079	0.009	0.084	0.005	0.085	0.006	0.085	0.006
DISTRAIL (300m)	0.130	0.039	0.133	0.030	0.132	0.035	0.135	0.028	0.137	0.025†	–	–
GEODIST	-0.061	0.120	-0.061	0.130	-0.072	0.085	-0.066	0.111†	–	–	–	–
DISTRIV (300m)	0.073	0.146	0.073	0.132	0.074	0.139†	–	–	–	–	–	–
URBCOV	0.065	0.141	0.067	0.133†	–	–	–	–	–	–	–	–
LOGSIZE	-0.042	0.230†										
R^2	0.3290		0.3272		0.3230		0.3178		0.3134		0.2949	
PROBABILITY OF R^2	0.0001		0.0001		0.0001		0.0001		0.0001		0.0001	

Table 6.4b: *Forward selection procedure for selecting an optimal subset of explanatory matrix variables for derelict site species assemblages. The model parameters (Std. b) are standard partial regression coefficients. All probabilities (P) are one-tailed, computed after 9999 random triple permutations of the dependent matrix variable; minimum probability for each model is therefore 1/(9999+1) = 0.0001; At each step the variable chosen for selection is marked with a asterisk (*); the variable with the smallest probability is selected if its probability is smaller than the Bonferroni-corrected significance level p = 0.10/{number of variables in the model at the given step}.*

MATRIX VARIABLE	STEP 1		STEP 2		STEP 3		STEP 4	
	Std. *b*	P	Std. *b*	P	Std. *b*	P	Std. *b*	P
ENVT	0.459	<0.001 *	–	–	–	–	–	–
SIMHAB (500m)	0.284	<0.001	0.258	<0.001 *	–	–	–	–
AGE	0.118	0.014	0.109	0.012	0.104	0.012	0.104	0.012 *
SUBS	0.176	<0.001	0.118	<0.001	0.101	0.002 *	–	–
DISTRAIL (300m)	0.184	0.014	0.148	0.025	0.133	0.030	0.133	0.032
GEODIST	-0.119	0.044	-0.090	0.057	-0.082	0.083	-0.079	0.080
DISTRIV (300m)	0.046	0.286	0.046	0.260	0.069	0.155	0.064	0.174
URBCOV	0.113	0.062	0.106	0.051	0.088	0.083	0.081	0.098
LOGSIZE	-0.026	0.345	-0.063	0.145	-0.060	0.143	-0.059	0.150
R^2	0.2104		0.2742		0.2842		0.2949	
PROBABILITY OF R^2	0.0001		0.0001		0.0001		0.0001	

Mantel correlations (ENVT, SUBS, SIMHAB and AGE). The forward selection output demonstrates that environment alone can account for the majority of variation explained by the model. The increment of the R^2 attributable to proximity of similar habitat, the only landscape component included, is low (0.06) but nevertheless significant, suggesting that sites in high density derelict land areas tend to have a larger than expected number of 'shared' species. There is no evidence from these analyses to support the hypothesis that rail or river corridors are important in determining species composition at derelict sites.

(ii) Floristic diversity

The best predictors of different measures of site diversity, as identified by stepwise linear regression modelling, are shown in Table 6.5. None of the landscape variables (urban land cover, derelict land cover nor measures of connectivity) proved significant explanatory variables for any of the different measures of diversity used in these analyses.

Both total species number (TOTRICH) and the number of native species (NATIVERICH) were found to show strong positive relationships to site size ($p<0.001$ in both cases). Sites on old arable land and those over sandy substrates were also found to be less diverse. At the quadrat level (QUADRICH), very little variation between sites could be explained by the variables tested, although sites over brick rubble had slightly less species per quadrat. However brick rubble, ballast and basic conditions were all positive predictors of mean specialism (MEANSPEC). The numbers of species found at a site which were recorded as locally uncommon by the Ecorecord database (RARERICH) was positively associated with both site area and less fertile sites, with only fertility remaining a significant component when these were counted as a percentage of the total assemblage.

(iii) Representation of traits

The best predictors of the representation of species traits, as identified by stepwise linear regression modelling, are shown in Table 6.6. The best model of mean seed weight of species (SEEDWEIGHT) included negative associations with moisture, salinity and fertility and the presence of ballast as the underlying substrate. This environmental model was significantly improved ($t= -4.12$, $p<0.001$) by the inclusion of URB100, indicating that seed weight was lower at sites that were immediately surrounded by a high percentage of concrete.

Table 6.5: *Summary of stepwise multiple regression analysis of site variables on different measures of diversity showing significant predictors. Log transformation of data set is indicated where used. Variables relating to urban land cover, derelict land density and corridor distances were also included in the model but provided no significant predictors.*

	STEPWISE MODEL				LOGSIZE		ELLENBERG ENVIRONMENT VARIABLES		SUBSTRATE VARIABLES	
	R^2	d.f.	F-value	P-level	*t*	P	*t*	P	*t*	P
TOTRICH	0.416	3,46	10.940	<0.001	5.02	**** LOG			AGRICTOP - 0.31 SAND - 0.25	** ***
NATIVERICH	0.303	2,47	10.193	<0.001	4.35	**** LOG			AGRICTOP -2.13	*
QUADRICH	0.094	1,48	4.963	0.031					BRICK -2.23	*
MEANSPEC	0.339	3,46	7.873	<0.001			REACTION 3.23	*** LOG	BALLAST 2.70 BRICK 2.17	* *
RARERICH	0.213	2,47	6.355	0.004	2.57	* LOG	NITROGEN -2.06	* LOG		
%RARE	0.200	2,47	5.872	0.005			NITROGEN -3.24	* LOG		

Table 6.6: *Summary of stepwise multiple regression analysis of site variables on plant traits showing significant predictors. Log transformation of data set is indicated where used. Site age, Site area and corridor distances were also included in the model but provided no significant predictors.*

	Stepwise model				Site Age		Ellenberg Environment Variables			Substrate types		Urban cover		Distance to corridors	
	R^2	d.f.	F-value	P-level	*t*	*p*	*t*	p		t	p	*t*	p	*t*	p
Seedweight	0.671	5,44	17.94	<0.001			Moisture -4.24 Salinity -2.61 Nitrogen -2.52	**** * LOG *		Ballast -6.23	****	Urb100 -4.12	****		
Seedlongev	0.563	4,45	14.47	<0.001	-4.39	****	Reaction 5.88	****				Urb1000 -2.48	*	Distrail -3.16	***
Ruderality	0.663	4,45	22.11	<0.001			Light 7.76 Nitrogen 3.78	**** LOG ****		Conctar 2.89	**	Urb5000 -3.25	***		
%Zoochor	0.480	2,47	7.04	0.002			Light -2.96	***		Conctar -2.63	*				

The best model of mean seed longevity (SEEDLONGEV) showed longevity to be higher on more basic substrates and on younger sites. It was also found to be greater at those within close proximity to a rail corridor (DISTRAIL, $t=-3.16$, $p<0.005$). The model was slightly improved by the inclusion of a negative association with urban land cover at intermediate radius (URB1000, $t=-2.48$, $p<0.05$).

The best model for the ruderality of the species assemblage of a site (RUDERALITY) showed ruderality to be considerably higher on more open light sites with high fertility. More ruderals tended to occur on sites over broken tarmac and at sites which at the wider radius had less urban cover (URB5000).

Species with zoochorous adaptations were shown to contribute a smaller proportion of a sites flora in more open sites and on sites over tarmac.

(iv) Individual species distributions

Species that were significantly more frequently recorded ($p<0.1$) at sites closer to rail corridors are listed in Table 6.7. These species are without exception those that may be commonly be found either on the rail ballast or otherwise growing on rail embankments. Furthermore, of the 16 species, 12 are alien species and only 4 natives. This compares to 108 aliens found in the survey as a whole out of a total species list of 379 and represents a significantly higher than randomly expected number of aliens ($\chi^2=16.33$, $p<0.001$). Both pre-1500 archaeophyte and post-1500 neophyte aliens are well represented in this subset but notably 4 neophytes are represented amongst the 8 species showing highest affinity to rail track. A smaller number of species were found to be significantly associated with sites closer to river corridors, listed in Table 6.8. Again aliens were represented in larger numbers than would be expected by chance ($\chi^2=3.12$, $p=0.078$). All four alien species were archaeophytes.

As would be expected with the significance of similar habitat (SIMHAB) in the similarity models, a much greater number of species were positively associated with sites found close to another derelict land patch (Table 6.9). Alien species were once again represented in high numbers (9 of the 13 most significant associations) with both archaeophyte and neophyte species included. Furthermore, 11 of the 35 species were found to be members of the typically heavy-seeded Fabaceae, compared to only 28 amongst the total species list, a greater number than would be expected by chance alone ($\chi^2=21.70$, $p<0.001$).

Table 6.7: *Results of one tailed t-Tests investigating proximity of sites to nearest railway line and the likelihood of a species presence. Species listed were more likely to occur at sites closer to a railway. (**** p<0.001; *** p<0.005; ** p<0.01; * p<0.05; † p<0.10).*

Species	t-value	d.f.	p-value	Rail corridor habitat	Status
Cytisus scoparius	4.10	40	****	track / bank	Neophyte
Linaria vulgaris	2.93	46	***	track	Native
Aster novi-belgii	3.09	19	**	bank	Neophyte
Crepis vesicaria	2.89	18	*	track	Neophyte
Artemisia absinthium	2.59	35	*	track / bank	Archaeophyte
Melilotus officinalis	2.57	42	*	track	Archaeophyte
Lathyrus latifolius	2.59	14	*	bank	Neophyte
Urtica dioica	2.87	7	*	bank	Native
Artemisia vulgaris	2.29	17	*	track / bank	Archaeophyte
Papaver rhoeas	2.23	16	*	track	Archaeophyte
Foeniculum vulgare	2.30	9	*	track	Archaeophyte
Lapsana communis	2.05	24	†	track	Native
Lamium purpureum	2.13	7	†	track	Archaeophyte
Verbascum thapsus	2.11	7	†	track	Native
Lamium album	1.86	20	†	bank	Archaeophyte
Lupinus x regalis	1.79	12	†	bank	Neophyte

Table 6.8: *Results of two tailed t-Tests investigating proximity of sites to nearest river corridor and the likelihood of a species presence. Species listed were more likely to occur at sites closer to a river. (**** p<0.001; *** p<0.005; ** p<0.01; * p<0.05; † p<0.10).*

Species	t-value	d.f.	p-value	Status
Persicaria maculosa	3.33	45	****	Native
Senecio vulgaris	2.88	36	**	Native
Tripleurospermum inodorum	2.51	41	*	Archaeophyte
Chelidonium majus	2.63	11	*	Archaeophyte
Polgonum aviculare	2.33	41	*	Native
Rumex obtusifolius	2.87	5	*	Native
Poa annua	1.96	44	†	Native
Lamium album	1.71	8	†	Archaeophyte
Salix fragilis	1.02	9	†	Archaeophyte

Table 6.9: *Results of two tailed t-Tests investigating proximity of sites to nearest derelict site and the likelihood of a species presence. Species listed were more likely to occur at sites closer to another derelict land site. (**** $p<0.001$; *** $p<0.005$; ** $p<0.01$; * $p<0.05$; † $p<0.10$). Asterisk by species indicates member of Fabaceae.*

Species	t-value	d.f.	p-level	Status
*Melilotus officinalis**	3.51	40	****	Archaeophyte
Linaria vulgaris	4.38	34	****	Native
Foeniculum vulgare	4.39	47	****	Archaeophyte
Centaurium erythraea	3.73	36	****	Native
*Lupinus x regalis**	3.48	31	****	Neophyte
Vulpia myuros	4.38	47	****	Archaeophyte
Artemisia absinthium	3.69	28	****	Archaeophyte
Hieracium agg.	3.6	46	***	Native
*Trifolium dubium**	3.03	41	***	Native
Reseda luteola	3.18	33	***	Archaeophyte
Malva sylvestris	2.84	45	**	Archaeophyte
Oenothera glazioviana	2.92	33	**	Neophyte
Buddleja davidii	1.82	35	**	Neophyte
*Lotus corniculatus**	2.6	27	*	Native
Tragopogon pratenis	2.43	46	*	Native
Rumex crispus	2.53	22	*	Native
*Vicia sativa**	2.64	16	*	Native
*Vicia cracca**	2.37	42	*	Native
Tussilago farfara	2.53	17	*	Native
Solanum dulcamara	2.38	24	*	Native
Hypericum perforatum	2.2	41	*	Native
Centaurea nigra	2.17	41	*	Native
Populus tremula	2.14	42	*	Native
Senecio squalidus	2.14	24	*	Neophyte
Poa trivialis	2.06	39	*	Native
*Ulex europaeus**	1.97	37	†	Native
Festuca rubra	2.22	8	†	Native
*Trifolium arvense**	1.9	29	†	Native
Leucanthem x superbum	2.02	12	†	Neophyte
*Cytisus scoparius**	1.86	35	†	Native
*Trifolium pratense**	1.88	14	†	Native
Aster novi-belgii	1.8	24	†	Neophyte
Cynosurus cristatus	1.76	47	†	Native
*Medicago lupulina**	2.46	3	†	Native
Leucanthemum vulgare	1.74	33	†	Native

6.5 DISCUSSION

Floristic composition

The results indicated that the landscape variables tested in general accounted for very little of the variation which occurs between the species compositions found on derelict sites in the West Midlands. The age of the site, substrate and environmental conditions give the greatest indication of the form of the assemblage that is likely to be encountered. However the models also showed that there was considerable variation in the data set that remained unaccounted for. This unexplained variation is probably largely attributable to two factors: (i) the haphazard disturbances and input of material which is both characteristic of derelict sites and which gives them their diversity (Plates 6.1 and 6.2); and (ii) the very complex nature and diversity of the surrounding land mosaic.

(i) Disturbance and tipping

As previously discussed in Chapters 3 and 4, disturbance is an inherent and unpredictable factor influencing the flora of sites. These events may alter the species composition dramatically, particularly by reopening the vegetation for pioneer species on older sites. Where tipping of waste occurs, the vegetation becomes even harder to predict because many casual garden plants may become established, which would otherwise have no potential to reach a site. In addition, garden weed seeds may germinate and proliferate on the heaps of soil which are deposited. The vegetation at the edges of sites, where these events are most likely to occur, can be viewed as especially unpredictable.

(ii) Complexity of the urban landscape mosaic

Compared to natural landscapes, or indeed rural landscapes, the urban zone has an incredible diversity of land uses and features. It is simply not feasible to translate all this information into quantitative measures and it is equally hard to infer which measures may be of greatest relevance to the dispersal of plants. Within this study the importance generally of urban cover, derelict cover and railways and rivers have been considered. However other factors which may be important to determining species composition of a site may be the diversity and

Plate 6.1: *Burnt out cars at Vincent drive. Small scale disturbances such as these are common events on derelict land sites in the West Midlands.*

Plate 6.2: *Bin bags of garden waste dumped at the edge of a derelict site in Kings Heath, West Midlands. Dumping of large quantities of soil is also common and may lead to distinctive mounds appearing on sites each with distinct vegetation from the rest of the site.*

amounts of other surrounding habitat types and other linear features such as roads and walls. In addition measures could have been taken not from surrounding radii of sites but from an arc towards the direction of the prevailing wind. It is not therefore demonstrably possible to rule out a much larger effect of landscape on recruitment and it can only be conclusively stated that no evidence of a significant relationship was found here.

Diversity

Existing theories about species diversity can account for the majority of the variation that exists between sites in their species richness. Larger sites typically held more species as would be expected under the species-area relationship (MacArthur & Wilson 1967). Very high fertility was shown to account for low diversity in quadrats, as predicted by several theories of resource availability versus species richness which predict low diversity at extremes (e.g. Odum 1963, Grime 1973). This may also account for the poor species numbers recorded at agricultural sites (high fertility) and sandy sites (low fertility and low water availability).

Increased specialism may also have been expected on substrates containing a high number of basic ions as these sites produce analogous conditions to those on semi-natural chalk and limestone grassland situations, enabling specialist species typical of these habitats, such as *Sanguisorba minor*, *Carduus nutans* and *Anthyllis vulneraria* to thrive. There were no highly acidic sites and consequently the relationship described by the data appears unidirectional.

No clear relationship was shown between diversity measures and any landscape variable, suggesting the factors considered were of little importance to total site diversity. Once again, due to the considerable unexplained variation we cannot rule out an effect of other landscape variables having an impact on diversity.

Traits

Environment again appears to be the major determining factor influencing plant trait composition on a site. In particular, as might be expected from other studies (e.g. Donelan & Thomson 1980; Pyšek, Prach & Šmilauer 1993; see also Chapter 4), seed longevity reflects site age. Additionally, the closely correlated trait seed weight (Thompson, Band & Hodgson 1993), responds to fertility and moisture, which were themselves demonstrated to be related

to successional age on derelict sites in Chapter 3.

There was no direct evidence to suggest increased isolation had a major effect on the representation of plant traits related to dispersal at these sites. There are at least two major reasons that may account for this.

Firstly, the majority of derelict land species, particularly on younger sites, are well adapted to dispersal anyway and they may therefore reach sites with little difficulty. For the minority of immobile species, the scales at which fragmentation and isolation were measured in this study (i.e. 100m to 5km) may be irrelevant if only natural recolonisation is considered to be important. To a woodland specialist, for example, a minimum distance of 500 metre or 5 kilometres may essentially be equally effective barriers to population interchange (Kirby & Thomas 1994). Indeed, several studies have shown that the level at which isolation becomes significant is difficult to demonstrate convincingly (Connor & McCoy 1979; Hart & Horowitz 1991), suggesting that most species fall either into the highly competent or the very poor disperser groups (Dawson 1994b).

A second but not mutually exclusive reason why heavier-seeded species do not appear to respond to site isolation is that a large number of these species may grow successfully in the intervening landscape matrix. Many tree and shrub species found on derelict land (such as *Corylus avellana*, *Quercus robur* and *Cotoneaster* species) are widely planted or grow as freely propagated individuals across the urban landscape, in street plantings, ornamental beds, and in green spaces, such as parkland and gardens. It is clearly very difficult to ascertain geographical distance between populations of these species and consequently their sensitivity to isolation cannot be easily gauged.

There is, however, some limited indirect evidence from the trait results to suggest a link to dispersal pathways and species present at sites. Increased seed longevity at sites nearer railways may be indicative of the greater resistance of these species to spraying measures which take place to reduce vegetation on these linear features. The greater tendency for lighter seeds to occur on saline locations may also reflect dispersal of seeds by motor vehicles to the edges of derelict sites beside roads where salinity may be higher (Jones 1981). Lighter-seeded species are known to be better represented in mud transported by vehicles (Hodkinson & Thompson 1997).

Individual species distributions

The tendency for many heavy-seeded legume species to be found only close to other potential source habitats is perhaps the clearest indicator from this study of a spatial landscape affect on species composition. The explanation of why this structuring is so clear in this group and not in other heavy-seeded taxa is not clear however. Many legumes are important early colonisers of derelict land favoured by their mycorrhizal associations with nitrogen fixing bacteria (Gilbert 1989). It is possible that some of these species are particularly dependent on the window of opportunity offered by a freshly denuded site, after which time they are excluded by the existing vegetation. Consequently, speed of arrival may be particularly significant and the benefit of being located closer to a source population may be particularly high for these species. Activities, such as the movement of bulldozers from one demolition site to another and between sites with existing vegetation being cleared for new development, may be particularly crucial for some of these heavy-seeded species.

While corridors did not explain significant amounts of variation recorded between site assemblages in the West Midlands, there is evidence from this study to suggest that the presence of corridors may have an impact on the distribution of a small number of species. This seems to be primarily as source habitats in their own right, as the few species which show a significant tendency to occur near both rail and river are particularly well adapted for proliferation in the respective habitats offered by the corridor itself. These adaptations can be defined by traits such as the seed longevity of railtrack species (e.g. *Melilotus officinalis*) enabling herbicide spraying tolerance; and the low-growing life forms of river corridor species (e.g. *Poa annua* and *Polygonum aviculare*) enabling tolerance to trampling on these recreational routes. Adaptation can also be even more easily inferred simply by the abundant occurrence of all these species on the corridors to which they show positive association.

Of course, the potential for linear proliferation and subsequently transportation to isolated patches in areas adjacent to forward reaches of corridor means that, with time, a dispersal effect could be argued for these species. However, in the absence of a true conduit function, linear habitats should be viewed solely as to the extent to which they complement the spatial arrangement of other habitat patches. Conduit function itself assumes that dispersal in the linear space will be accelerated, compared to within an area of similarly homogenous habitat

with no obvious potentially directional structure (Forman & Godrun 1984). Railtrack is known to act as a conduit for *Chaenorrhinum minus* (Arnold 1981) because the vortices produced by trains enhance the dispersal potential of its seed (presumably the conduit function disappears for a disused railway however!). *Chaenorrhinum minus* was not common enough on West Midlands derelict sites to suggest whether it showed any spatial structuring as a result of this benefit. However, a number of species that did show a positive association with sites proximity to rail corridors were light seeded species with dispersal adaptations (e.g. *Crepis vesicaria*, *Artemisia absinthium*), which may potentially benefit in a similar manner. However, this characteristic also suggests that these species should in reality be amongst those least in need of assistance in traversing the urban matrix. Several species in Table 6.7 have seeds whose terminal velocity (*sensu* Askew *et al.* 1997) is considerably higher than that for *Chaenorrhinum minus* (Table 6.10) and for which the vortex benefit seems unlikely. This is particularly so for the heavier-seeded legumes *Lupinus x regalis* and *Lathyrus pratensis*, which appear to be **predominantly** embankment species on West Midland rail corridors and so do not come into very close contact with trains. The majority of these larger-seeded species are known to be dispersed by animals and there is certainly evidence to suggest that mammals use urban corridors as foraging routes (Yalden 1980, Page 1981). However, it is also the case

Table 6.10: *Comparison of values for seed terminal velocity index (Ken Thompson* unpublished data*) between species significantly more frequent near railways and* Chaenorrhinum minus. *Red indicates species with seeds that undergo faster descent. Blue indicates seeds that undergo slower descent.(Indicates best estimate where no data available)*

SPECIES	TERMINAL VELOCITY INDEX (Estimate if n/a)
Chaenorrhinum minus	**1.70**
Cytisus scoparius	n/a (>5.0)
Linaria vulgaris	2.25
Aster novi-belgii	0.33
Crepis vesicaria	0.40
Artemisia absinthium	n/a (0.45)
Melilotus officinalis	3.79
Lathyrus latifolius	n/a (>5.0)
Urtica dioica	2.44
Artemisia vulgaris	0.45
Papaver rhoeas	2.14
Foeniculum vulgare	3.28

that the most frequent urban animals (e.g. fox, *Vulpes vulpes*, wood mouse, *Apodemus sylvaticus,* grey squirrel, *Sciurus,* and brown rat *Rattus norvegicus*) are typically not significantly restrained by, for example, a suburban area, as most residents in these areas can testify.

The fact that much fewer species were found to be significantly associated with sites near to river corridors may be indicative of the much greater disparity between the kind of habitats offered predominantly by these particular linear features and the habitats found on derelict land sites in the conurbation. In addition, discontinuities in habitat connectivity tend to be much greater along urban rivers. This can be due to interruptions caused by culverts, bridges and channelisation that limit riverbank vegetation or also due to the tendency for vegetation to vary incorporating recreational grassland, coarse grassland, scrub and woodland along the course of the corridor. The species which have been found here to have an affinity with sites near river corridors include species with zoochorous adaptations (e.g. *Lamium album* and *Rumex obtusifolius*), hydrochorous species (e.g.. *Salix fragilis*) and species tolerant of trampling pressure (e.g. *Polygonum aviculare* and *Persicaria maculosa*). As with species in Table 6.7, these are plants that are well adapted to survival within the corridor with which they are associated, and not surprisingly they are also abundantly recorded in along river corridors as well as the derelict sites situated near them. It is true that further direct research on seed dispersal would be needed to properly test whether corridors really do function as conduits for the few plant species that showed spatial affinity to them. Unfortunately, these studies involve a time period commitment which is often beyond the scope of modern research programmes.

From an urban conservation perspective, the results of this study indicate that, if corridors are effective, then it is only for a very limited number of species whose status within the conurbation is by no means threatened. Indeed, the significantly high proportion of alien species within these lists suggests that rather than preserving native biodiversity, urban corridors may be far more influential at abetting the spread of immigrant species. There is certainly historical evidence in the UK which suggests these linear features may be important pathways for increasing national distribution (e.g. Kent 1960, 1964). This issue is discussed further in Chapter 7.

6.6 CONCLUSIONS

a) The geographical location of a derelict site within the West Midlands conurbation and its spatial proximity to other derelict sites and linear habitats have been shown to have much less of an influence on species composition than the environmental conditions which prevail. The age since denudation and the nature of the remnant substrate after demolition have both been shown to be particularly important factors influencing species present;

b) The species richness of a site was also shown to be independent of any geographical attributes tested here. Established theories on patterns of richness under different site size and resource availability scenarios explained the most variation;

c) There was also no significant decline in the number of heavy-seeded species at sites that were isolated from other derelict land or from corridors. However, the greater number of species with high seed longevity near to railtrack and with lighter seeds on more saline sites may reflect some degree of dispersal along railways and roads respectively;

d) Species that showed closest affinity to sites adjacent to linear habitats were also those that were typically frequently found also growing on these corridors and are well adapted for survival and dispersal along these routes. However, none of these species were particularly uncommon in the urban zone and most appear well adapted to disperse across the urban mosaic in general;

e) Amongst individual species of heavy-seeded taxa, there was little relationship between distribution and the proximity of potential source habitats. The distinct exception to this was exhibited by members of the Fabaceae, many of which were only found in high density derelict areas with close neighbouring derelict flora. The movement of construction vehicles to and from these sites may be important for this group to become established while the short window open to pioneer species remains;

f) In general the results indicate that the spatial arrangement of sites as tested is of only limited consequence to the floristic similarity and has no effect at all on the species diversity exhibited. Where patterns may emerge in simple datasets, these may instead

be largely be attributed to spatial correlation between key environmental circumstances and geographical location rather than to recruitment limitation effects. It is suggested firstly this may be attributable to most species falling on either side of a dispersal potential divide. Under this scenario, the vast majority are either easily capable of transferring regenerative propagules across the distances applicable in this study or are so sensitive to isolation that even distances well within the connectivity thresholds utilised here are effectively barriers to dispersal. Secondly it is pointed out that within the complex urban mosaic, many species can survive in a multitude of locations such as gardens, planted areas, walls, untended verges and cracks in pavements. Under these circumstances, it is truly difficult to measure true isolation for a large number of taxa and consequently difficult to test dispersal-related hypotheses effectively.

CHAPTER SEVEN

AN INVESTIGATION INTO SPATIAL PATTERNS IN THE OCCURRENCE OF ALIEN PLANTS ON DERELICT LAND ACROSS AN URBAN ZONE, THE WEST MIDLANDS, UK.

SUMMARY

Cities often represent the first point of arrival for many immigrant plant species. It is therefore desirable to develop a greater understanding of the establishment and spread of alien plants in urban areas. In the last few decades, the alien and native floras of several central European cities have been subjected to quantitative comparison. In contrast work on this field in the UK has been limited. This paper utilises floristic data both from a recent survey of 50 derelict land plots in the West Midlands, UK and from regional records to investigate patterns of alien richness in relation to urban land cover measures and other features of the urban landscape. A distinction is made between pre-1500 AD immigrants (archaeophytes) and post-1500 AD introductions (neophytes), and differences in their distribution are examined. A total of 384 higher plant species were recorded of which 256 (67%) were native, 77 (20%) were neophytes and 51 (13%) were archaeophytes. Aliens in general were found to be positively correlated with immediate measures of site urbanity. However this relationship was most clearly shown by neophytes. Differences in the frequency and distribution of archaeophyte and neophyte taxa were found to be closely linked to the timing of their introduction and their relative ecological amplitude. The findings suggested that many neophyte taxa may be better adapted to invade semi-natural and natural habitats than archaeophyte species and that the full effect of this may be subject to a considerable time delay. The removal of environmental constraints due to climate change and nitrogen deposition may increase the cause for conservation concern.

KEYWORDS: *Aliens, Neophytes, Archaeophytes, Urban ecology.*

7.1 INTRODUCTION

The distribution of alien species within the UK has received considerable attention in recent decades. The majority of these studies have focused on the spread of invasive species along riparian zones (e.g. Perrins, Fitter & Williamson 1993), while there are also a few well documented accounts of the spread of a few species in or along anthropogenic habitats (e.g. Kent 1960, 1964) and much debate concerned about the characteristics that make a successful invader (e.g. Crawley, Harvey & Purvis 1996; Hodkinson & Thompson 1997). There has, however been little comprehensive research on alien species specifically in urban areas of Britain, although Gilbert's (1989) work identified aliens that were particularly typical for several cities. Nevertheless, the distribution and spread of alien plants across cities is of particular interest for they are the point of origin for the majority of newly arriving alien species (Sukopp & Werner 1983). In contrast to Britain, a number of major studies have been carried out in central Europe which have examined the contribution of alien species to the flora of cities (e.g. Sukopp *et al.* 1979; Sukopp & Werner 1983; Kowarik 1985; Pyšek 1998). Work in this region has also examined the spatial patterns of alien distribution (e.g. Kowarik 1990, Pyšek 1995) and enabled quantitative comparisons between cities (e.g. Kunick 1982; Pyšek 1998) and consequently the identification of consistent trends in alien distribution, as will be discussed below.

Contribution of aliens to the richness of the urban flora

In general, the contribution of aliens to urban areas is high compared to natural or semi-natural outlying habitats (Crawley 1987). Many urban floras in continental Europe have very high alien components (>40%) and have been recorded as having higher total species than their surrounding landscapes (Walters 1970; Haeupler 1974; Kornas 1978; Sukopp & Werner 1983; Kowarik 1985; Pyšek 1993). However, recent analysis comparing urban and rural tetrads in the UK has suggested that urban land cover does not equate to higher species richness as any gain in alien species is balanced by a loss in natives (Roy, Hill & Rothery 1999). These authors have questioned the validity of conclusions from central European studies on the basis that under-recording in rural areas may have occurred.

Archaeophytes and neophytes

The simple division of species into alien and native is not adequate to interpret patterns of alien arrival, persistence and spread. There are considerable differences between the manner of arrival and habitat requirements of species introduced due to human activity now and in the past. Between the end of the last Ice Age and well into the Middle Ages, the majority of introductions associated with human activities occurred as a result of the transport of arable seed. Deliberate introductions for crops or herbal use were probably outnumbered, however, by the accidental introduction of weeds that came with them. The numbers of immigrant species arriving in any particular time period is likely to have varied, depending particularly on political and economic factors. The Roman conquest, in particular, is associated with the arrival of many new species to the British flora (e.g. *Aegopodium podagraria*) (Rackham 1986). A number of these species became abundant but some have also subsequently become rare with the introduction of highly intensive farming methods (e.g. *Agrostemma githago*) or in the case of some parasitic species due to the discontinuation of growing certain crops e.g. (*Orobanche ramosa*) on hemp. During the last five hundred years, agricultural introductions have continued, but increasingly escapes from cultivated gardens or ornamental beds has become more significant. Some species, though native, may have also benefited from subsequent reintroductions from these sources (e.g. *Campanula trachelium*), and in many instances the provenance of these plants may also be foreign. The rapid spread of some the most successful of these latterly introduced species (e.g. *Senecio squalidus*, *Matricaria discoidea* and *Buddleja davidii*) demonstrates the benefit of being able to exploit the new niches offered within a developed and urbanised region. In central Europe, the classification of aliens according to time of origin is well-established (Holub & Jirasek 1967). This classification has enabled comparison between archaeophyte (pre-1500 introductions) and neophytes (later introductions) when examining the distribution of aliens in cities. In a comparative study of European urban floras, Pyšek (1998) found that neophytes became a more significant component of the alien flora compared to archaeophytes as city size increased. It is likely that the spatial distribution and relative abundance of archaeophytes and neophytes within cities will reflect the different time periods and places of establishment and, where applicable, the different representation of ecological attributes within the groups.

Aims of the study

The aim of this study is to provide an overview of the over-riding patterns of alien species distribution on derelict land in the West Midlands and to infer where possible if these patterns are applicable for the region as whole. It is also intended to answer the specific questions:

(i) What is the contribution of alien species to the richness and diversity of the urban derelict flora in the West Midlands?

(ii) Which characteristics of the landscape, if any, determine the contribution of alien flora to a particular derelict location?

(iii) Is the relative abundance of neophyte and archaeophyte taxa also influenced by these factors?

(iv) To what extent does the current distribution of alien species reflect the manner and timing of their introduction and can predictions be made about the future spread of alien taxa from this?

7.2 METHODS

Floristic data

The floristic data utilised were from a survey of 50 derelict land sites across the West Midlands conurbation as described in detail in Chapter 6 (see also Appendix I and Figure 6.1). Species lists were drawn up for higher plants at each site (Appendix IX). Species records for the region as a whole were obtained from *Ecorecord*, the wildlife database for Birmingham and the Black Country, and are derived from Nature Conservancy Council surveys in the area conducted between 1985-1992. These data are given in Appendix VIII. Nomenclature follows Stace (1997). Classification of some species as alien or native is subject to debate. In this study native, archaeophyte or neophyte status is attributed according to Preston, Pearman & Dines (*in press*). The combined data sets were used to derive the number and proportion of native, alien, archaeophyte and neophyte species at each site. Information on the habitat types into which each alien species has naturalised was taken from Stace (1997).

Measurements of landscape characteristics

Measures of the land cover surrounding each site were quantified at three distances from the site boundaries (100m, 1km and 5km). The land cover within the 100m was mapped on the ground between August-October 1999. Each map was digitised, and incorporated into an ArcView GIS (ESRI, Copyright 1992-1999, Version 3.2). ArcView was then used to extract the percentage of concrete and percentage of gardens within each buffer (CONC100, GARD100, Appendix Xa). At the wider scales, the Land Cover Map of Great Britain (LCMGB, Fuller *et al.* 1994) was used. This map is derived from satellite imagery (with a 25m grid resolution). Percentages of urban land cover within 1km and 5km buffers around each site (URB1000, URB5000, Appendix Xb) were then calculated in ArcInfo. ArcView was also used to provide accurate measures of site area, measured in square metres (SIZE).

Data from the Joint Development Trust (JDT) (1998) database of derelict land in the West Midlands was added to the ArcView GIS, and the shortest distance in kilometres between each site and a neighbouring derelict patch was extracted (DISTSIM, Appendix Xc). The proximity of each site to the nearest railway and nearest river were measured in metres using Ordnance Survey maps (DISTRAIL, DISTRIV, Appendix Xc). The OS maps were also used to establish the boundary of the conurbation, and the shortest distance between site and the urban edge was measured in kilometres (DISTEDGE, Appendix Xc).

Estimates of site age (AGE) were established on the basis of responses to questionnaires, the JDT database and old city maps as described in Chapter 3 and Appendix IV.

7.3 RESULTS

The contribution of alien species to the urban flora in the West Midlands

Of the 384 plant species recorded at the 50 sites studied, 256 (67%) were classified as native and 128 (33%) were classified as alien, of which 51 (13% of total species) were archaeophytes and 77 (20% of total species) were neophytes (Figure 7.1). The twenty most frequently recorded alien species found in the derelict land survey are listed in Table 7.1 and include 13 archaeophyte and 7 neophyte species. The most widespread neophyte species were *Acer pseudoplatanus*, *Buddleja davidii* and *Senecio squalidus*.

Figure 7.1: *Proportion of native and alien species in the total list of 374 species found in a survey of 50 derelict sites.*

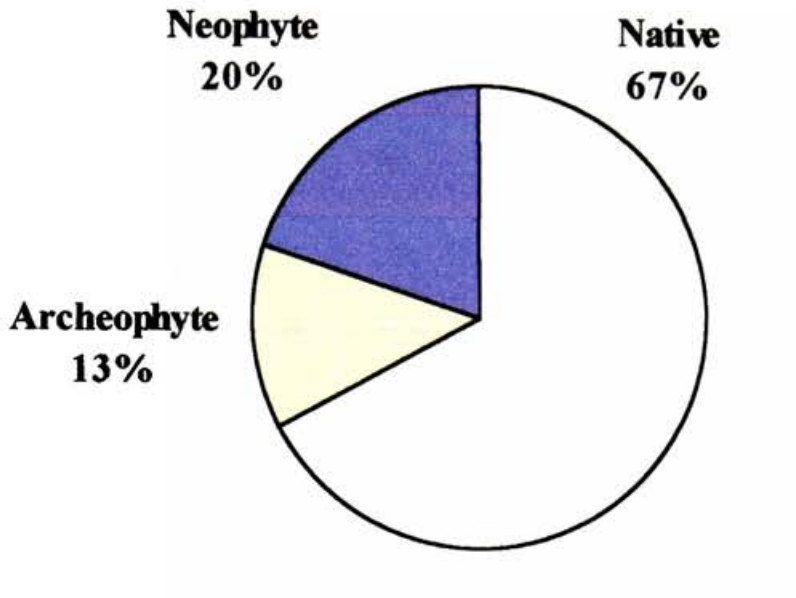


Table 7.1: *The twenty most frequently recorded alien species in the survey*

	NUMBER OF SITES	ARCHAEOPHYTE / NEOPHYTE	INFORMATION REGARDING INTRODUCTION (Clapham, Tutin & Moore 1989 unless otherwise noted)
Artemisia vulgaris	37	Archaeophyte	Unknown
Acer pseudoplatamus	30	Neophyte	Native in C or E Europe, introduced in early C16th
Senecio squalidus	30	Neophyte	From C or S Europe, escaped from Oxford Botanic Garden 1794.
Tripleurospermum inodorum	28	Archaeophyte	Unknown. Native in continental Europe & W. Asia.
Artemisia absinthium	27	Archaeophyte	Unknown
Geranium dissectum	26	Archaeophyte	Unknown
Buddleja davidii	25	Neophyte	Introduced from gardens c.1890. Native of China
Anisantha sterilis	25	Archaeophyte	Unknown
Reseda luteola	25	Archaeophyte	Unknown
Lactuca serriola	22	Archaeophyte	Probably from SW Asia or Siberia. Long cultivated as a salad plant
Sisymbrium officinale	21	Archaeophyte	Unknown
Solidago canadensis	19	Archaeophyte	Planted from 1648 (Mabey 1997) from New World. First recorded as naturalised in C19th.
Melilotus officinalis	18	Archaeophyte	Unknown
Conyza canadensis	17	Neophyte	Introduced 1690 (Mabey 1997), possibly in the contents of a stuffed bird (Clement & Foster 1994)
Hordeum murinum	16	Archaeophyte	Unknown
Epilobium ciliatum	15	Neophyte	Timber alien from N. America. First record 1891
Armoracia rusticana	14	Archaeophyte	Probably SE Europe & W Asia. Introduced, relic of former cultivation
Fallopia japonica	12	Neophyte	Introduced in 1886 (Stace 1997)
Silene latifolia	12	Archaeophyte	Probably introduced in neolithic times
Linaria purpurea	11	Neophyte	Introduced from Sicily (Clement & Foster 1994)

Figure 7.2: *Plot of number derelict sites where found vs. number of regional records for native, archaeophyte and neophyte species.*

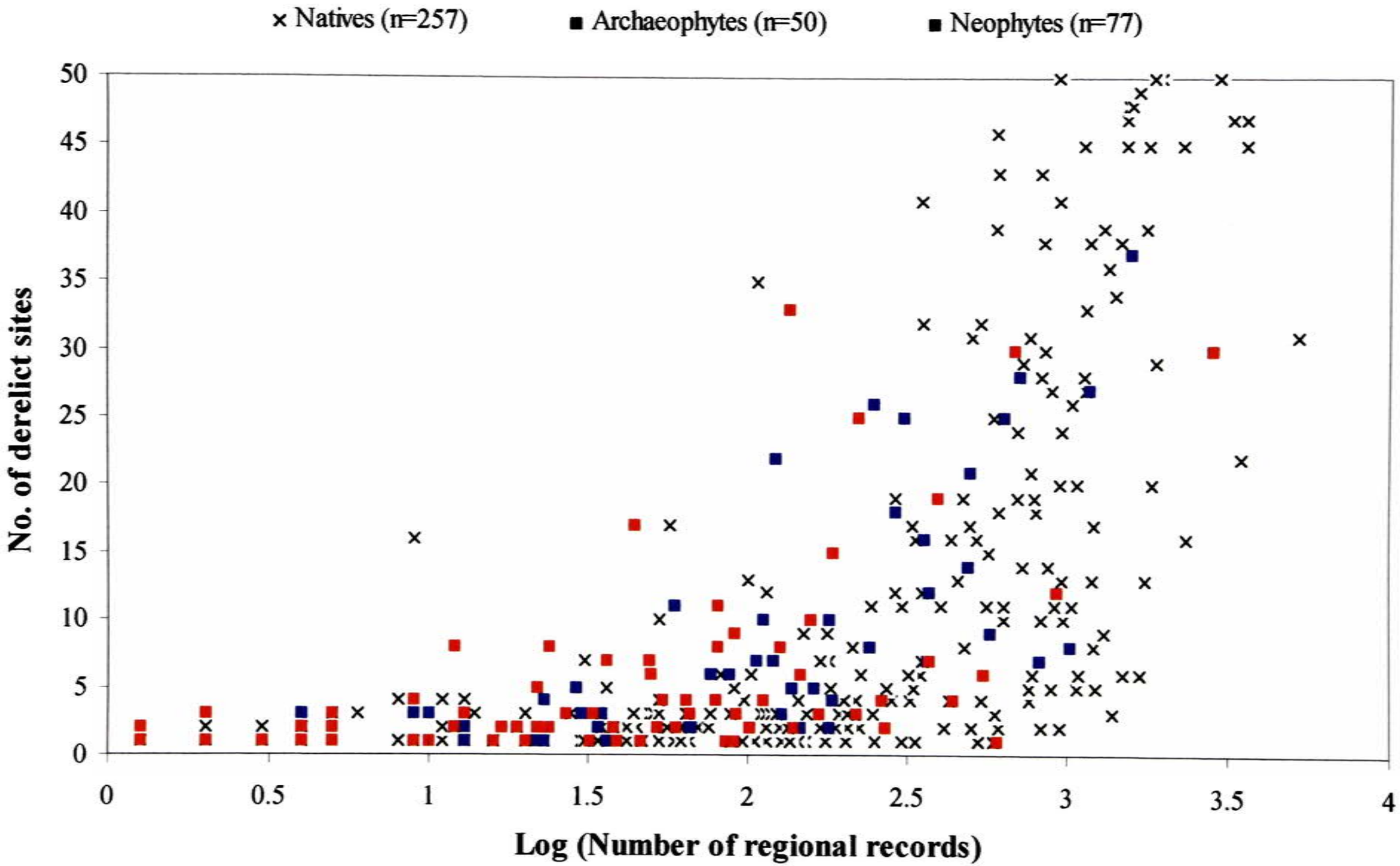


Table 7.2: *Mann-Whitney U tests of the differences in the recorded frequencies of native and alien species, using data from the derelict survey and from the regional Ecorecord database.*

	DERELICT SURVEY	ECORECORD
Native species vs. All alien species	W = 48576.5, $p < 0.05$	W = 53035.0, $p < 0.0001$
Native species vs. Archaeophytes	W = 37026.5, n.s.	W = 38705.5, $p<0.005$
Native species vs. Neophytes	W = 42426.0, $p < 0.005$	W = 45205.5, $p < 0.0001$
Neophytes vs. Archaeophytes	W = 3839.0, $p < 0.005$	W = 3949.0, $p < 0.001$

Table 7.3: *Chi-square tests on the number of archaeophyte and neophyte species that according to Stace (1997) have naturalised into at least one semi-natural habitat.*

	ARCHAEOPHYTE SPECIES	NEOPHYTE SPECIES
Number regularly naturalised into at least one semi-natural or natural habitat	5	24
Number not regularly naturalised into any semi-natural or natural habitats	46	53
Total	51	77
$\chi^2 = 7.99$, df = 1, p = 0.005		

The patterns of occurrence exhibited by species across these groups varied considerably. Figure 7.2 demonstrates that the species which were both most frequently observed in this study and most abundantly recorded in this region in all habitats were predominantly natives. Of the species that were found in at least 20 of the survey sites and with over 500 records in the area, 89% were native. Neophyte species were generally poorly distributed with only 5.2% of species being found at more than a quarter of derelict sites and with 200 regional records, compared to 27.6% of natives and 20.0% of archaeophytes.

Differences in species frequencies for the subsets of natives, neophytes, archaeophytes and all aliens found in the derelict land survey were tested using non-parametric Mann Whitney U tests (Table 7.2). Native species on average showed higher frequency than alien species as a whole and particularly compared to neophyte species both across the derelict sites and in the region as a whole. Interestingly, archaeophyte species showed similar frequency patterns to the native subset across derelict sites but in the regional Ecorecord data set, the native subset still showed higher frequency patterns on average. This suggests that archaeophyte species were poorly represented in the semi-natural and natural habitat patches contributing to the city's floristic records.

The habitat types into which each alien species has naturalised was determined from Stace (1997), and categorised as either highly anthropogenic (waste land, cultivated land, waysides, rough ground or walls) or semi-natural (scrub, woodland, river banks, wetlands or dunes) habitats. The "grassland" and "hedge" categories were excluded from consideration, since differentiation between anthropogenic or semi-natural habitat was particularly arbitrary. Chi-square tests (Table 7.3) showed that a significantly greater proportion of neophytes had become naturalised in semi-natural habitats while archaeophytes, despite their much longer period of establishment were more confined to anthropogenic habitats.

The effects of landscape characteristics on the alien flora

As a preliminary analysis, all floristic variables and all landscape variables were tested for normality using Kolgomorov-Smirnov tests. All conformed to normal distributions with the exception of SIZE ($Z = 1.638$, $p = 0.009$), which was normally distributed when transformed to log site size; URB5000 ($Z = 1.51$, $p = 0.021$) which was normally distributed when cubed ($Z = 1.17$, $p=0.132$) and DISTSIM ($Z = 1.526$, $p = 0.019$), which was normally distributed when the square root was taken ($Z = 0.722$, $p = 0.674$). All landscape variables (including those that were transformed) were then correlated with each other using two-tailed Pearson Product Moment Correlation. The resulting matrix is given in Table 7.4 and shows that many of the landscape variables were strongly correlated with each other. Each of the floristic variables was correlated with each of the landscape variables (Table 7.5), and then stepwise linear multiple regression modelling was used to identify the best predictors of each floristic variable. The results are given in Table 7.6.

The results from Table 7.5 and Table 7.6 revealed that native species richness was strongly positively correlated with site size and was also best predicted by LOGSIZE alone. The number of native species showed no significant responses to any other of the landscape measures. These findings were also demonstrated in Chapter 6. There was a non-significant trend of increasing native richness towards the city centre (DISTEDGE, $r = 0.25$, $P<0.1$ Table 7.5) but this was most likely attributable to the positive correlation which existed between the size of the sites surveyed and their proximity to the urban centre (Table 7.4).

The number of archaeophyte species was also positively related to site size and was best predicted by LOGSIZE alone (Table 7.6). Archaeophyte richness was also found to be

Table 7.4: *Correlation matrix between site variables demonstrating autocorrelation in the data set (Two-tailed Pearson's r value and significance indicated). ns not significant, * $p<0.05$, ** $p<0.01$, *** $p<0.005$, **** $p<0.001$.*

	URB100								
GARD100	-0.187 ns	GARD100							
URB1000	**0.415** ***	0.071 ns	URB1000						
URB5000 (cubed)	**0.283** *	-0.064 ns	**0.778** ****	URB5000 (cubed)					
DISTSIM (sqrt)	**-0.330** *	0.164 ns	**-0.513** ****	**-0.441** ***	DISTSIM (sqrt)				
DISTRAIL	-0.246 ns	0.107 ns	**-0.517** ****	**-0.434** ***	**0.340** *	DISTRAIL			
DISTRIV	0.134 ns	0.224 ns	0.149 ns	**0.340** *	-0.182 ns	0.147 ns	DISTRIV		
DISTEDGE	**0.281** *	-0.090 ns	**0.679** ****	**0.852** ****	**-0.558** ****	**-0.456** ***	**0.312** *	DISTEDGE	
LOGSIZE	0.006 ns	**-0.341** *	0.077 ns	0.216 ns	-0.238 ns	-0.102 ns	0.094 ns	**0.358** *	LOGSIZE
AGE	**-0.437** ***	-0.079 ns	-0.046 ns	0.058 ns	-0.129 ns	-0.108 ns	0.230 ns	0.039 ns	-0.031 ns

Table 7.5: *Correlation matrix between the relative composition of different immigrant categories in the derelict flora (y) and the landscape variables (x). Correlation coefficients (n=50) and their significance levels are shown. Models providing the best fit are indicated. L=linear (y=a+bX), P=power ($y=a^{X}$), E=exponential (y=exp(a+bX)) and LOG (y=.log(a+bX)*
*ns not significant, † p<0.10, * p<0.05, ** p<0.01, *** p<0.005, **** p<0.001.*

	URB5000 (cubed)	URB1000	DISTEDGE	URB100	URBINDEX	DISTRAIL	DISTRIV	DISTSIM (sqrt)	GARD100	LOGSIZE	AGE
NUMBER OF NATIVES	ns	ns	**0.25 † L**	ns	**-0.27 † LOG**	ns	ns	ns	ns	**0.48 *** L**	ns
NUMBER OF ALIENS	**0.27 † L**	**0.40 *** L**	**0.33 * L**	**0.36 * P**	**0.61 **** P**	**-0.26 † LOG**	ns	**-0.24 † L**	ns	**0.36 * L**	ns
NUMBER OF ARCHAEOPHYTES	ns	**0.25 † L**	ns	**0.25 † E**	**0.69 **** P**	**-0.31 * LOG**	ns	**-0.24 † P**	ns	**0.36 ** L**	ns
NUMBER OF NEOPHYTES	**0.37 ** P**	**0.49 ****LOG**	**0.40 *** P**	**0.36 * E**	**0.40 *** P**	ns	**0.25 † L**	**-0.26 * LOG**	ns	**0.27 † L**	ns
PROPORTION OF ALIENS	ns	**0.42 *** L**	ns	**0.42 *** L**	**0.75 **** E**	**-0.27 † LOG**	ns	ns	ns	ns	ns
PROPORTION OF ARCHAEOPHYTES	ns	ns	ns	**0.26 † L**	**0.74 **** P**	**-0.31 * LOG**	**-0.26 † LOG**	ns	ns	ns	ns
PROPORTION OF NEOPHYTES	**0.29 * L**	**0.51 **** L**	**0.29 * L**	**0.42 *** E**	**0.55 **** P**	ns	**0.30 * E**	ns	**0.24 † L**	ns	ns
RATIO NEOPHYTES TO ARCHAEOPHYTES	**0.43 *** P**	**0.38 ** P**	**0.43 *** P**	ns	**-0.30 † L**	ns	**0.39 ** P**	ns	**0.29 * L**	ns	ns

Table 7.6: *Linear multiple regression models of the relative composition of different immigrant categories in the derelict urban flora (y) by landscape variables (x). Stepwise analysis performed using $p < 0.05$ to include and $p > 0.10$ to remove.*

	VARIABLES IN MODEL	SIGNIFICANCE OF MODEL
NUMBER OF NATIVES	LOGSIZE: t= 3.79, p<0.001	r^2=0.230, df=1, 48. F=14.35, p<0.001
NUMBER OF ALIENS	URB1000: t= 3.03, p<0.004; LOGSIZE: t= 2.67, p<0.011	r^2=0.273, df=2, 47. F=8.81, p=0.001
NUMBER OF ARCHAEOPHYTES	LOGSIZE: t= 2.71, p=0.009	r^2=0.132, df=1, 48. F=7.32, p=0.009
NUMBER OF NEOPHYTES	URB1000: t= 3.47, p=0.001; LOGSIZE: t= 2.79, p=0.008; LOGGARD: t= 2.66, p=0.011	r^2=0.364, df=3, 46. F=8.77, p<0.001
PROPORTION OF ALIENS	LOGURB100: t= 3.38, p=0.001	r^2=0.192, df=1, 48. F=11.41, p=0.001
PROPORTION OF ARCHAEOPHYTES	DISTRAIL: t= -2.62, p=0.007; LOGAGE: t= -1.99, p=0.049; LOGDISTEDGE: t= -1.79, p=0.05	r^2=0.191, df=3, 46. F=3.68, p=0.033
PROPORTION OF NEOPHYTES	URB1000: t= 4.70, p<0.001; LOGDISTRAIL: t= 2.05, p=0.046	r^2 =0.324, df=2, 47. F=11.27, p<0.001
RATIO OF NEOPHYTES TO ARCHAEOPHYTES	LOGDISTEDGE: t= 3.24, p=0.002; DISTRAIL: t= 2.18, p=0.035; GARD100: t= 2.04, p=0.047	r^2=0.259, df=3, 46. F=5.36, p=0.003

significantly higher close to railways (DISTRAIL, -0.31, $p<0.05$ Table 7.5), although this variable was not included in the model (Table 7.6) because of some correlation between DISTRAIL and LOGSIZE (Table 7.5). However, the proportion of archaeophytes in the site flora *was* best modelled by DISTRAIL, with the model being further improved by with the addition of LOGAGE and LOGDISTEDGE, although this model did not perform very well in general ($r^2=0.135$, $F=3.68$, $p=0.033$, Table 7.6). The relationship between railways and the abundance of archaeophytes was also shown by the results of individual species distributions in Chapter 6. Indeed,43.8% of those species showing a significant relationship were archaeophytes.

The number of neophyte species was most strongly related to urban cover (URB1000, $r=0.51$, $p<0.005$, Table 7.5). The best predictive model found for neophyte richness included URB1000, LOGSIZE and LOGGARD ($r^2=0.364$, $F=8.77$, $p<0.001$, Table 7.6) indicating a greater richness not only at the more urban and larger sites, but also at those site surrounded by a greater proportion of gardens. The proportion of neophytes was also best predicted by the surrounding urban cover (URB1000), but the model also included a negative association with railways (LOGDISTRAIL), which is likely to be due to the positive association of archaeophytes to this variable.

On grouping archaeophytes and neophytes together, the total number of aliens was best predicted by a combination of urban cover and site size (URB1000 and LOGSIZE, $r^2=0.273$, $F=8.81$, $p=0.001$ Table 7.6), which reflected the best predictors for neophyte and archaeophyte richness respectively. The proportion of aliens was best modelled by urban cover at the local scale (LOGURB100, $r^2=0.192$, $F=11.41$, $p=0.001$ Table 7.6), although URB1000 was itself strongly correlated with other measures of urban cover at the wider scales (Table 7.5). The ratio of neophytes to archaeophytes was found to be best modelled by LOGDISTEDGE, DISTRAIL and GARD100, with archaeophytes being more frequent towards the urban edge and near to railways, while neophytes were higher towards the city centre and where a high density of gardens surrounded the site.

7.4 DISCUSSION

Summary of the findings

If the distribution patterns of natives, neophytes and archaeophytes recorded for the West Midlands are compared to those summarised for central European cities by Pyšek *et al.* (1998) several consistent trends are observed. To summarise: (i) the proportion of aliens is higher in more urban regions; (ii) in particular, the proportion of neophytes is higher towards the urban centre. Pyšek *et al.* 1998 found a greater proportion of neophytes in larger cities which can be viewed as a similar result); and (iii) neophytes on average occur at lower frequency than both archaeophytes and natives.

In addition, this study has identified that compared to the native species found on derelict sites, archaeophytes were more poorly distributed in remnant natural and semi-natural habitat fragments and that their restriction primarily to anthropogenic habitats in the West Midlands reflects their recorded distribution UK-wide (*as in* Stace 1997). It was also shown that the few neophyte species that are abundantly distributed in cities all have a long period of establishment (i.e. at least over 100 years) and that the wealth of habitats that neophytes in general have been shown to exploit in the UK is significantly wider than for archaeophytes despite their shorter presence here. The results from the previous chapter (Table 6.5) emphasised there was no significant relationship between species richness and proximity to the urban centre on derelict sites despite the increase numbers of neophytes. The only influence of surrounding habitats was to alter the ratio observed between archaeophytes and neophytes, with the former being better represented at sites near railways, while the latter became more significant where garden densities was higher.

Aliens in cities

The preponderance of aliens in more urban zones, and indeed their generally large contribution to urban flora, which has been shown here is in keeping with cities across the world. Pyšek *et al.* (1998) identified the importance of these urban zones as sources of alien immigration and the differences in the environment compared to outlying regions as the major factors which contribute to this. Although in this study the number of native species

was not shown to decline significantly towards the urban centre, only one habitat type was considered. Generally the natives that are lost along a rural-urban gradient are those that are representative of the more pristine semi-natural and natural habitats, which either become mostly small and highly fragmented or disappear altogether in the urban zone (Roy, Hill & Rothery 1999). Consequently, any gain in neophyte species within the West Midlands conurbation compared to the rural districts which surround may be counterbalanced by the loss of natives although the net result is debatable. Indeed, the question of whether overall species richness increases or decreases with urbanisation is perhaps rather an impossible question to answer due to the difficulty in establishing which aliens are truly naturalised and which may merely be persistent casuals.

Explaining the different patterns of archaeophyte and neophyte distribution across the urban landscape

The results from this study that demonstrate the importance of railways in determining the proportion of archaeophytes and neophytes are open to misinterpretation. Many archaeophyte species could certainly be well pre-adapted to these xerophytic free-draining habitats, as these conditions may be analogous to those encountered in their places of origin, which in many cases was probably disturbed southern European habitats. However, as was demonstrated also in Chapter 6, the distribution of several neophyte species (notably *Cytisus scoparius*, *Aster novi-belgii*, *Crepis vesicaria* and *Lathyrus latifolius*) proved to be closely linked to these linear features. In addition, the well-documented spread of many notable recent introductions both here in the UK (e.g. *Senecio squalidus,* Kent 1960, 1964) and in continental Europe (e.g. *Heracleum mantegazzanium,* Otte and Franke 1998; and *Senecio inaequidens*, Ernst 1998) has highlighted the significance of railways as possible dispersal pathways for neophytes.

It seems therefore that the result indicates a greater proportional significance of railways to the archaeophyte group as a whole, when compared to neophytes, of which many more have proven capable of persisting in semi-natural habitats (Table 7.3). This apparent difference in ecological amplitude can probably be explained by the significant difference in the manner in which species were 'selected', accidentally or otherwise, for introduction.

Archaeophyte species were predominantly introduced accidentally as weed seeds amongst crop materials or deliberately as crops. Many demonstrate a common suite of quite specific

functional attributes which are beneficial for occurring within the traditional arable niches e.g. persistent seed bank, high seed production capacity. In contrast, many neophytes have been introduced deliberately by collectors visiting overseas because of their botanical or horticultural interest. Accidental introductions have also continued to occur but the types of materials (e.g. aggregates) in which they are transported have become much more diverse than simply foodstuffs. These later neophyte introductions therefore represent a much more random and diverse selection of functional attributes and include species from a broader range of natural native habitats. Consequently, many invasive neophyte species may not form significant persistent seed banks (e.g. *Impatiens glandulifera*) or indeed produce fertile seed in the UK at all (e.g. *Fallopia japonica*). Instead a range of other attributes gives them the ability to proliferate. This also means that there is a greater diversity of opportunity in terms of habitats for the spread of neophytes as a group and may explain their reduced spatial clustering around habitats which may be analogous to natural circumstances for archaeophytes.

Further evidence to support this theory is provided by the results in Table 7.2, which demonstrate that, while archaeophytes as a group have proven adept at exploiting the anthropogenic derelict sites in the West Midlands, they have shown much poorer integration into other habitat types compared to native derelict species. Indeed, from a conservation viewpoint, there is not much evidence to suggest any significant deleterious affect on native flora from the introduction of the vast majority of these long established species. The invasive species that have been the attention of the most research in the UK and northern Europe are predominantly all neophytes forming large dominant stands (e.g. *Impatiens glandulifera*, *Heracleum mantegazzanium*, *Fallopia japonica* and *Rhododendron ponticum*).

However, despite their greater flexibility, taken as a group, many neophytes are only found at very low densities (Tables 7.1-2). Two possible reasons can help explain this finding. The first explanation is that many neophyte species recorded on derelict land sites in this survey were ephemerophytes persisting on piles of garden refuse or indeed as remnant plantings in former gardens around which the land and houses have been bulldozed. The likelihood for many of these species to spread and become truly naturalised is often poor as climatic conditions may prevent regular seed set, while their vegetative means of spread may not be efficient enough to overcome the barriers to dispersal which they face. This is also probably

the major factor contributing to the inclusion of surrounding gardens within the best model for predicting the proportion of neophytes (Table 7.6), as these ephemerophyte species are typically close to habitation or source population. It also explains why neophytes are recorded at significantly lower numbers in less developed areas (Table 7.6), where residential gardens, ornamental plantings and municipal parks tend to be at much lower density.

The second explanation of neophytes occurring typically at lower frequencies across the conurbation, however, is that there are also a significant number of neophyte species for which the potential for expansion has not yet been fully realised. The evidence to support this is indirect and comes rather from consideration of those aliens which are abundantly distributed (Table 7.1). Of the 12 most frequent aliens only 3 were neophytes and these three all have a long history of introduction. The most frequent *Acer pseudoplatanus* has a very long history in the British Isles with most estimates suggesting around the beginning of the 16th century, placing it only just within the neophyte category anyway. The next most abundant neophyte on the list was *Senecio squalidus*, which has been present for over 200 years. Finally, the third of these most frequent neophytes, *Buddleja davidii*, was well known in gardens before the 19th century ended. The other neophyte species in Table 7.1 have all also been present for over a century at least.

Implications for the future alien flora of the UK

The suggestion from these findings therefore is that a substantial period of establishment is essential for newly arrived aliens to become truly ubiquitous, even if they are able to distribute far and wide quickly. Studies on invasive tree species have also identified that those which cause the greatest problems are generally those that have been planted most widely and for the longest time (Richardson 1998).

Ultimately, the speed with which a potentially successful alien species will become widespread is depended on its own unique set of characteristics (Williamson & Fitter 1996) and even highly successful invaders will vary in the time taken to expand their populations. Nevertheless, the proliferation of the few neophyte species mentioned may only be a preview of an inevitable and much larger invasion, when it is considered how many new species have only been planted widely in the last century.

Of even greater concern is that this spread may not be restricted to primarily human

influenced habitats but that the characteristics that many of these species have will enable them to encroach into natural communities in manner far outweighing the impact of the majority of archaeophyte species. In addition, changing environmental factors may further benefit future invasions. The medium term effects of global warming on the UK climate may be enough to remove temperature restraints which can prevent the spread of a species and limit them to central urban zones only (e.g. *Ailanthus altissima* in Berlin, Sukopp & Weiler 1983). Increased atmospheric nitrogen deposition may also disrupt natural ecosystems, making them more vulnerable to invasion (Pyšek & Pyšek 1995). It seems probable therefore that the significance of the threat to native plant conservation caused by alien taxa may be set to rise.

7.5 CONCLUSIONS

a) The distribution of alien species in the West Midlands show patterns consistent with trends observed in central Europe. The proportion of alien species is highest in areas of highest urban cover and this can mostly be explained by an increase in post-1500, neophyte species.

b) Neophyte species tended on average to be found at much lower frequency across the derelict survey sites than both archaeophyte and native species. Some neophyte species are abundant, however, and these are species with long periods of establishment.

c) Despite being well distributed across derelict sites, most archaeophytes are not common in other habitats in the urban zone. This corresponds closely with their recorded status in the UK flora, which demonstrates they are closely linked to anthropogenic habitats. In contrast, neophyte species are recorded as naturalised from a more diverse range of habitats in the UK. Consequently, they are also the alien taxa most likely to pose a conservation threat.

d) The differences between archaeophytes and neophytes and their relative amplitudes for habitat exploitation can largely be put down to historical and cultural differences in the manner in which species were introduced.

e) The implications of the fact that many neophytes will be better adapted to invade natural habitats and that the full extent of their spread may not be reached for at least several centuries after their introduction may have dramatic consequences for the British flora. The effects may be heightened by the removal of traditionally occurring physiological constraints to their spread with climate change and nitrogen deposition.

CHAPTER EIGHT

SPATIAL VARIATION WITHIN URBAN DERELICT SITES: A STUDY OF SEED RAIN, VEGETATION AND SEED BANK

SUMMARY

Higher species diversity at the edges of habitats has been widely observed. Four possible explanations of this phenomenon are that: (i) edges recruit a wider source pool of species; (ii) greater environmental stress at the edge prevents competitive dominance; (iii) disturbance events prevent competitive dominance at the edge; and (iv) complementarity of resources occurs at the boundary of two habitats.

A comparative study of higher plant diversity was undertaken on site edges and interiors on 3 derelict land sites in the West Midlands, UK. A seed rain trapping method was employed to measure deposition of seed. Ellenberg values and plant traits were utilised as indirect measures of environment and disturbance.

Greater species richness at the edge was recorded at all sites. Seed rain was highly patchy and was dominated by a few species abundantly distributed in the immediate vegetation. Species unique to site edges were poorly represented in the seed rain. Species present at edge patches exhibited no greater variance in Ellenberg values than those found within site interiors. Few significant differences were found between the mean Ellenberg values for species in edges and interiors. A higher than expected representation of annual and persistent-seeded species was found amongst species unique to site edges.

The results provide little support for a dispersal-driven mass effects explanation or for either complementarity of resources or increased environmental stress as factors

maintaining higher diversity at the edge. The findings of this study suggest that higher species diversity at the edges of derelict sites is driven primarily by a higher intensity of disturbance which enables the regeneration of annual species suppressed during succession in the site interiors. The possible significance of anthropogenic dispersal and pollution is also discussed.

KEYWORDS: *Edge effect, Seed rain, Sinks, Ellenberg values, Disturbance, Urban ecology*

8.1 INTRODUCTION

Higher species diversity within the edges of habitat patches is a widely reported feature for many taxa including plants (Harper 1977; Ranney, Bruner & Levenson 1981; Lovejoy *et al.* 1984; Quinn & Robinson 1987). Gilbert's (1989) observations of derelict land also acknowledged that plant species richness could be particularly high on site perimeters. The distance from the perimeter for which an edge effect is recorded will vary depending on the taxa being observed and the habitat types that meet at the boundary. Species typical of edge habitats may also be found in the interior and many interior species will also occur towards the edge; however there will be decreasing or increasing gradients of abundance respectively across the perimeter-interior divide (Bastin 1997).

Proposed explanations for higher species numbers at site boundaries

The observation of increased diversity at habitat edges has generated much discussion and several possible causal mechanisms have been proposed. In general, the arguments supporting one mechanism over another have their foundations in a much more long-standing ecological debate on how high species diversity is maintained when the essential requirements of plant species are so similar. In the study of edge effects, the explanations can be assigned to two general groups: those that identify deposition of seed as the primary factor, and those that attribute greater importance to environmental factors.

(i) Seed deposition:

It has been suggested that in a mosaic landscape of habitat (or micro-habitat) patches, local

diversity may be maintained by immigration from nearby sources (Shmida & Ellner 1984; Shmida & Wilson 1985). Such immigration may enable species to persist within a patch despite having a negative population growth rate, resulting in higher than expected levels of diversity. The concept of sink plant populations is usually applied at a wider landscape or even regional scale (e.g. Kellman 1996; Dunning, Danielson & Pulliam 1992). However, since most species deposit the vast majority of their seed within a few metres of the parent plant (Verkaar 1990), the concept of sinks and sources are at least equally attributable for plants at a highly localised scale within habitat patches.

Kunin (1998) pointed out that this theory could also explain higher diversity at spatial boundaries such as habitat edges. The boundary between quite similar habitats that share a large proportion of species would not be expected to show greatly increased diversity, while extremely different habitats would exclude the possibility of species of both types co-occurring. On this basis, Kunin proposed that *mass effects* should have the greatest impact on species richness between habitats of intermediate difference to each other.

Some authors (Schoenewald-Cox & Bayless 1986; Wiens, Crawford & Gosz 1986; Gosz 1991; Wiens 1991; Forman & Moore 1991) have described site perimeters as analogous to cellular membranes, being differentially permeable to the dispersal of diaspores, and suggest that this 'filter-function' can explain the differences in diversity between the perimeter and the interior. Under this theory, increasing impermeability of the boundary will create a declining gradient of diversity from the edge inwards. To some extent these theories can be viewed as complementary, as both place emphasis on seed dispersal as the primary mechanism determining diversity.

(ii) Alternative means by which high diversity may be maintained

Other proposed explanations for how high diversity in communities (edge or not) is maintained are typically based on the means by which competitive dominant species are excluded from successfully monopolising the available pool of resources (Shmida & Ellner 1984). Grime (1973, 1979) concluded that this was most likely to occur within plant communities where levels of *environmental stress*, notably mineral deficiency in the soil or drought, are high. Alternatively, he proposed that activities preventing the maximum growth of competitors, such as mowing, grazing and burning, could have a similar effect when

occurring in moderation. Species richness would be expected to rise at the edge if there is increased stress or frequency of these other activities. However species richness would fall if conditions become too extreme so that only highly tolerant species could persist. This prediction fits closely with Odum's (1963) observation that greatest diversity occurs in the middle range.

The intermediate disturbance hypothesis (Connell 1978, 1979) can also be related to Grime's concepts. At the highest levels of disturbance, only species that can produce propagules quickly enough can persist; while in the absence of disturbance, competitors are not restricted and therefore become dominant. The disturbance hypothesis can be equally applied to the intensity and area of disturbance events.

Another possible explanation put forward to explain higher species richness at the edge is that the boundary between two habitats may offer the environmental attributes of both adjacent habitats and consequently can contain a shared pool of species from each. This situation has been described as *complementarity of resources* (Forman & Godron 1984, Yahner 1988).

Approaches to testing these hypotheses

(i)Seed deposition

If species exist in edges as sink populations, they must receive a regular supply of propagules from the adjoining habitats. Consequently, to determine whether permeability of the border is a factor determining species richness, it is necessary to identify how the seed and propagule deposition varies across the site.

The dispersal of seed from plant populations is one of the most neglected areas of demographic research (Willson 1992). While there is a long history for research on the soil seed banks of numerous habitats (reviewed in Warr *et al* 1992), it is widely accepted that there is a considerable discrepancy between seed deposited on a site and that recorded in seed banks. Post-dispersal seed predation (e.g. Castro *et al.* 1999; Alcantara *et al.* 2000) and differential rates of burial and persistence within the soil (Thompson, Band & Hodgson 1993) combine to alter the composition and abundance of the species recorded in the soil, compared to those being deposited by the vegetation above. To gain a true understanding of seed deposition patterns, it is therefore desirable to monitor directly the content of the seed rain

The majority of recent seed rain studies have been carried out to determine the factors that influence the restoration of species-rich communities on degraded sites after major disturbances or on former agricultural land. Research has sought to identify whether recruitment limitation or site environment is the major obstacle to successful restoration. Studies on regeneration of species-rich grassland (Kollman & Pirl 1995; Hutchings & Booth 1996; Schott & Hamburg 1997) have suggested that seed rain alone cannot enable successful restoration, even where the target source pool is adjacent to a site. Similar findings have also been recorded for forest regeneration (Clark, Macklin & Wood 1998; Holl 1999; Zimmerman, Pascarella & Aide 2000; Wijdeven & Kuzee 2000). This can be attributed to the fact that most studies have also shown the seed rain to be predominantly dominated by species in the immediate vegetation (Peart 1989; Jefferson & Usher 1989; Jiminez & Aremsto 1992; Poschlod & Jackel 1993; Urbanska, Erdt & Fattorini 1998; Jensen 1998; Molau & Larsson 2000). The species richness of the rain is closely determined by that of the vegetation (Schott & Hamburg 1997) and the similarity of the seed rain to the vegetation is therefore often high, although in many instances the rain is dominated by abundant numbers of seeds from a small number of species (Peart 1989; Hutchings & Booth 1996; Stocklin & Baumler 1996). Furthermore, where multiple traps have been used, seed deposition has been shown to be exceptionally patchy both spatially (Rabinowitz & Rapp 1980; Peart 1989; Zobel *et al.* 2000) and temporally (Houle 1998).

(ii) Indirect measurements of environment and disturbance

The direct measurement of environmental variables may demand impractical levels of sampling and testing for the time constraints often applied to contemporary research. Ellenberg's development of indicator values for the central European flora (Ellenberg 1979; Ellenberg 1988; Ellenberg *et al.* 1991) enabled the placement of species within arbitrary scales for light, pH, soil moisture, fertility and salinity according to the range they exhibit across all habitats. This enables indirect information about the environment of a site to be taken from the species found growing within it. Nevertheless the utility of these values in the UK has been offset by the often considerable differences in ecological ranges exhibited by species in Britain when compared to the continent. This problem is overcome by the recent recalculation of the original Ellenberg values for species' realised niches within the British Isles (Hill *et al.* 2000). It is, therefore, possible to use these measures to compare the

environments of habitat patches to help elucidate the causes of higher species richness at the edge. A comparison of the mean values exhibited by species at different sectors can provide information on whether environmental stress is greater at the edge. In addition, the variance in values can be examined to determine whether variance is greater at the edge, as might be expected if resource complementarity is occurring.

Plant characteristics may also be used to interpret the overriding environmental constraints to which a plant community is exposed. Where disturbance plays an important role in shaping the vegetation, it would be expected to find more annual species, as setting seed quickly becomes particularly advantageous (Salisbury 1942). This is particularly so if it is combined with the ability to produce a persistent seed bank which can provide insurance against instability and consequently these attributes are closely correlated in plant taxa (Thompson *et al.* 1998).

Aims of the study

The aims of this study are to investigate the patterns of species diversity observed in stands of vegetation within discrete habitat patches of derelict land in the West Midlands conurbation. In particular, this work sets out to identify whether higher species richness is associated with the boundaries of these sites and to assess generally whether there is any indication that diversity in vegetation stands is spatially correlated. A simple seed trapping technique is tested to investigate directly how localised dispersal events may play a role in determining observed species richness. The effectiveness of this technique is examined and the patterns of seed deposition are observed and related these to the vegetation growing above and the seed bank in the soil below. Also investigated is how certain plant traits vary between these three stages. In addition, Ellenberg Indicator values and plant life traits are utilised to look for any specific environmental and disturbance related effects on diversity and for differences in environmental heterogeneity between edges and interiors.

8.2 METHODS

Site selection

Three derelict land sites were chosen for the study. Each was located within a different urban zone. Site LBM (Large Bentley Mill) is found towards the rural urban edge and had previously been a sports field. It ceased to be maintained 20 years ago and has become a prime location for fly tipping of household waste, garden waste and large numbers of old tyres which are episodically set ablaze. The vegetation is consequently exceptionally patchy. Some well-developed trees and shrubs, mostly non-natives, are scattered across the site while garden casuals are abundant, particularly on the heaps of rubbish. Thick grass swards are present in some places but in others the grazing pressure of numerous rabbits combined with trampling by people walking over the site has led to the development of a short-cropped turf. The perimeter is adjacent to a major road cutting through an out of town development. Site VD (Vincent Drive) is towards the inner city on land used for a metal works and factories in the past. The patch initially became derelict 15 years ago but has been regularly set ablaze by local children who also use it as part of an unofficial motorcross circuit. The vegetation is dominated in many places by alien tall herbs such as *Solidago* spp. and *Lupinus x regalis* and by *Rubus fruticosus* but is very patchy with low growing grasses and herbs also frequent. The boundary of the patch is predominantly scrub woodland. Site WR (Wilson Road) is situated within the suburban zone and was previously residential housing, since demolished. The patch became derelict in early 1990s and is not subject to any major disturbance. The vegetation mostly consists of thick grassy swards in which *Festuca rubra* and *Agrostis stolonifera* are dominant. The site perimeter is adjacent to a small suburban road with residential gardens within 10 metres in places.

The areas within the sites chosen for study consisted were as follows:

LBM	Perimeter 100m x 10m ($1000m^{-2}$)	Centre 4 x (25m x 10m strips) ($1000m^{-2}$)
VD	Perimeter 120m x 10m ($1200m^{-2}$)	Centre 4x (30m x 10m strips) ($1200m^{-2}$)
WR	Perimeter 80m x 5m ($400m^{-2}$)	Centre 4 x (20m x 5m strips) ($400m^{-2}$)

Field surveys of vegetation

Twenty 1x1 metre quadrats were surveyed for higher plant species at both the perimeter patch and across all four central patches. Braun Blanquet abundance cover scores were given to the species present. Quadrats were aligned in a linear fashion within each sample area at a distance of 4 metres from each other. In addition, walking line transects at intervals of 1 metre were conducted to draw up an overall species list for each sample area. A species list was also drawn up using the same method for the habitat adjacent to the perimeter being studied. The survey area for the adjacent habitat extended from the shared edge to a distance of 25 metres for all sites. Total site species lists were taken from the results of a field survey carried out in 1998 (see Chapter 2 and Appendix IX).

Seed rain study

The seed traps utilised in the experiment were constructed using plastic funnels (25 cm diameter) attached to bamboo canes between 50-75 cm off the ground. To hold the seed, muslin bags were attached securely to the end of the funnels with elastic bands. 40 funnels were used at each site, corresponding in placement with the central point of the field survey quadrats (described above). The period of seed collection was between July 1st and 10th September 1999. New muslin bags were attached halfway through the survey period. Bags collected for analysis were placed in individual sealed and marked polythene bags and returned to the laboratory. Any bags that were damp were air dried in the laboratory before being replaced to the polythene bag and being placed into freezer storage (-4°C).

In March 2000, the bags were removed from the freezer and thawed. The contents of each bag was taken to the greenhouse and spread over one sample seed tray (15 x 20 cm) containing a layer of sterile potting compost to 1cm depth over sterile sand. Greenhouse conditions were maintained at 16 hour daylight and day temperatures controlled between 15-22°C. All samples were watered twice daily. The period allowed for germination was three months for each tray. Seedlings were recorded and removed on identification. A number of taxa that were difficult to separate at the seedling stage e.g. *Sonchus* spp., *Epilobium* spp. were grouped.

8.3 RESULTS

Comparison between perimeter and interior vegetation

The total species richness of the vegetation in perimeter and interior patches were compared (Table 8.1a). Total species number recorded was higher in the perimeter patches at all three sites surveyed but at no site did this difference represent a significant result. A significant edge effect was suggested however when the results from all three sites were considered together using a paired t-test ($p<0.05$). The mean species number per quadrat was also significantly higher for perimeter patches at sites 1 and 2 (Table 8.1b).

The species unique to either the perimeter or interior patch were compared with the species present in the adjacent habitat (Figure 8.1). The results showed clearly that the unique species in the perimeter were significantly more likely to be also present in the boundary. However, it is noticeable that at all three sites over half of species unique to the perimeter were not recorded for the boundary habitat. This suggests that dispersal from the boundary cannot account for all the difference recorded in species richness between the edge and the centre.

Table 8.1: *Comparison of the species richness of the perimeter and interior areas at each site*

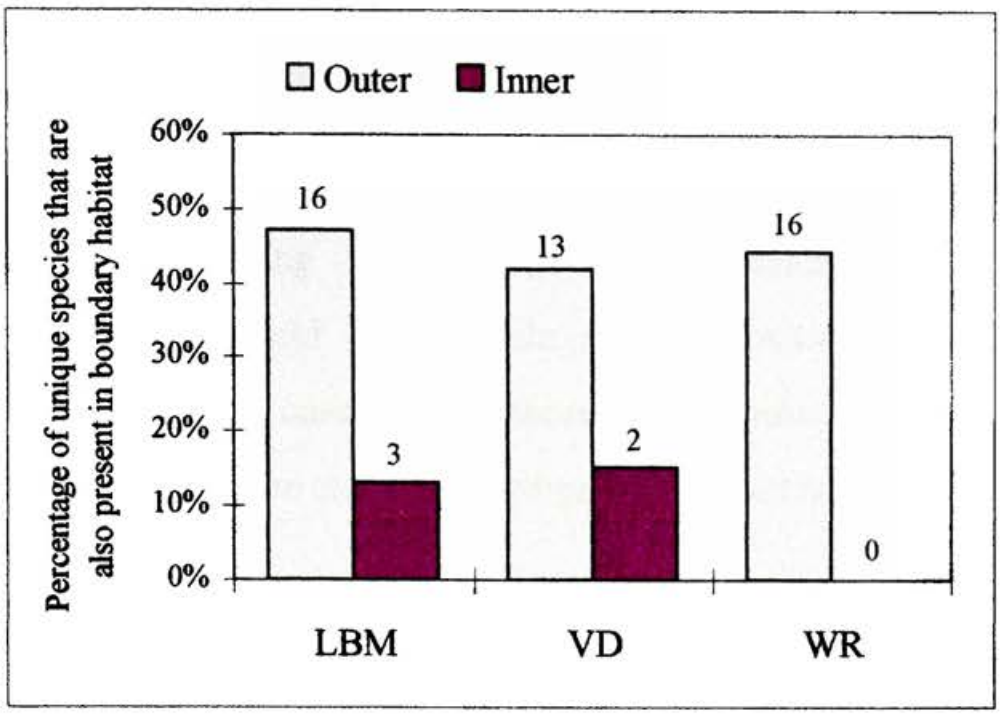
(a) Total species richness: Chi-squared and paired t-tests.

SITE	NO. SPECIES IN PERIMETER	NO. SPECIES IN INTERIOR	CHI-SQUARED TEST
LBM	92	80	$\chi^2= 0.42$, n.s.
VD	71	53	$\chi^2= 1.3$, n.s.
WR	79	53	$\chi^2= 2.6$, n.s.
			Paired T-test: $T= 4.6$, $p<0.05$

(b) Mean quarat species richness: Mann-Whitney tests.

SITE	QUAD RICHNESS: PERIMETER	QUAD RICHNESS: INTERIOR	MANN-WHITNEY TESTS:
LBM	7.8 (± 2.1)	6.9 (± 2.4)	p=0.067
VD	9.2 (± 3.5)	7.0 (± 1.9)	p=0.018
WR	7.8 (± 1.6)	8.4 (± 1.9)	n.s.

Figure 8.1: *The percentage of species unique to the perimeter and interior areas of the sites that are also present in the boundary habitat, and Chi-squared tests for significance*



	Chi-square Tests
LBM	$\chi^2 = 7.14$, p=0.0008
VD	$\chi^2 = 2.87$, p=0.09
WR	$\chi^2 = 6.82$, p=0.009

Characteristics of the seed rain

Despite attempting to reduce visibility of the funnel traps, it was always a risk within the urban area that traps would succumb to vandalism or accidental damage due to the high amount of recreational human activity at the sites. Recovery rates are summarised below:

LBM Bags from 28 funnels were successfully recovered (70% of total placed)

VD Bags from 31 funnels were successfully recovered (77.5% of total placed)

WR Bags from 36 funnels were successfully recovered (90% of total placed)

2,366 seedlings were germinated in the greenhouse representing 39 species or grouped taxa.

Table 8.2 lists the taxa recorded and their frequency and abundance in the seed rain at each site. The most widely distributed species was *Chamerion angustifolium,* for which germinable seeds were trapped in 56.5% of all funnels successfully recovered at the end of the study period. In contrast, this species was only found in 15.8% of the quadrats surveyed in the vegetation. The most abundantly recorded seeds were of *Urtica dioica* (984 seeds) and *Solidago* spp. (495 seeds) and their relative frequency in the rain and vegetation were very similar. *Urtica dioica* was recorded at 23.9% of funnels and at 27.5% of quadrats. *Solidago* spp. was recorded at 21.7% of both funnels and quadrats.

Figure 8.2a-c shows the abundance of seeds per funnel for the most common species at each site and illustrates that seed deposition occurred in an extremely patchy manner. This is also shown by the high deviation about the mean depositions given in Table 8.2. To investigate whether this observed patchiness was significant, the total pattern of deposition and the deposition rates of the most abundant species were tested for their goodness of fit to a Poisson distribution using Kolmogorov-Smirnov tests (Table 8.3). Conformity to a Poisson distribution would demonstrate seed deposition to be occurring in a random manner. However, in all cases tested, there was significant deviation from the expected distribution, demonstrating the extreme aggregation, or patchiness, of the deposited seed.

In Figure 8.3a-c, the species recorded in the seed rain are compared to the most abundant species recorded in the vegetation at each sample area and to the total species lists for each site taken from a previous study (see Chapter 6). This illustrates that the most abundant species in the vegetation account for most of the seed rain and that few species recorded in the rain were absent in the vegetation. Figure 8.3d-f compares the seed rain species to those recorded in the seed bank across the whole site (see Chapter 4). This shows that while many of the most abundant species in the seed bank were well represented in the seed rain, a considerable proportion of species found in the rain had not formed a persistent seed bank on the site.

Comparisons by some key plant traits illustrated that the species recorded in the seed rain showed much greater similarity to the flora than to the seed bank. Figures 8.4a-i and 8.5a-i demonstrate that the seed rain and vegetation have a higher representation of taller species and perennials. This was consistent across all three sites studied. Differences between the seed weight (Figure 8.6a-i) were less clear with the only noticeable trend being the absence or low frequency of very heavy seeded species from either the seed rain or seed bank. The results of one-way ANOVA and Tukey multiple comparison tests for trait representation are shown in Tables 8.4a-c.

Table 8.2: *Seed species found in the seed rain study, their abundance and mean catch (± standard error of the mean) at each site. Species sorted by number of funnels at which they were found.*

	ALL SURVEY		LARGE BENTLEY MILL (LBM)			VINCENT DRIVE (VD)			WILSON ROAD (WR)		
SPECIES	OCCURRENCE (% FUNNELS)	SEEDS FOUND	SITE FUNNELS	SITE TOTAL	MEAN WHERE FOUND	SITE FUNNELS	SITE TOTAL	MEAN WHERE FOUND	SITE FUNNELS	SITE TOTAL	MEAN WHERE FOUND
Chamerion angustifolium	56.5%	211	12	127	10.6 (±15.8)	16	29	1.8 (±0.9)	24	55	2.3 (±2.5)
Holcus lanatus	32.6%	202	2	2	1.0 (±0.0)	4	13	3.3 (±1.3)	24	187	7.8 (±11.1)
Urtica dioica	23.9%	984	11	659	59.9 (±90.0)	6	315	52.5 (±85.6)	5	10	2.0 (±1.7)
Solidago agg.	21.7%	495	0	0	0	12	270	22.5 (±35.7)	8	225	28.1 (±27.6)
Poa pratensis	21.7%	78	0	0	0	2	3	1.5 (±0.7)	18	75	4.2 (±6.4)
Epilobium agg.	20.7%	34	0	0	0	3	3	1.0 (±0.0)	16	31	1.9 (±1.8)
Senecio jacobaea	19.6%	80	10	48	4.8 (±4.6)	5	15	3.0 (±3.5)	3	17	5.7 (±5.0)
Agrostis stolonifera	17.4%	68	1	1	1.0 (±0.0)	3	42	14.0 (±22.5)	12	25	2.1 (±1.4)
Poa trivialis	10.9%	15	3	5	1.7 (±1.2)	0	0	0	7	10	1.4 (±1.1)
Sonchus agg.	8.7%	27	8	27	3.4 (±3.7)	0	0	0	0	0	0
Arrhenatherum elatius	8.7%	9	5	6	1.2 (±0.4)	1	1	1.0 (±0.0)	2	2	1.0 (±0.0)
Festuca rubra	7.6%	30	1	2	2.0 (±0.0)	1	1	1.0 (±0.0)	5	27	5.4 (±5.7)
Lolium perenne	7.6%	13	0	0	0	1	2	2.0 (±0.0)	6	11	1.8 (±0.8)
Betula pendula	7.6%	7	0	0	0	4	4	1.0 (±0.0)	3	3	1.0 (±0.0)
Juncus agg.	6.5%	14	1	1	1.0 (±0.0)	1	1	1.0 (±0.0)	4	12	3.0 (±2.2)
Alnus glutinosa	6.5%	6	1	1	1.0 (±0.0)	2	2	1.0 (±0.0)	3	3	1.0 (±0.0)
Poa annua	54%	21	2	15	7.5 (±9.2)	3	6	2.0 (±1.0)	0	0	0
Dactylis glomerata	5.4%	8	0	0	0	5	8	1.6 (±1.3)	0	0	0
Vicia hirta	4.3%	4	0	0	0	0	0	0	4	4	1.0 (±0.0)
Cerastium fontanum	3.3%	4	0	0	0	1	1	1.0 (±0.0)	2	3	1.5 (±0.7)
Rumex obtusifolius	3.3%	3	0	0	0	1	1	1.0 (±0.0)	2	2	1.0 (±0.0)
Scleranthus annua	3.3%	3	1	1	1.0 (±0.0)	1	1	1.0 (±0.0)	1	1	1.0 (±0.0)
Rumex acetosella	2.2%	29	2	29	14.5 (±19.1)	0	0	0	0	0	0
Galium aparine	2.2%	3	0	0	0	1	2	2.0 (±0.0)	1	1	1.0 (±0.0)
Lapsana communis	2.2%	3	0	0	0	2	3	1.5 (±0.7)	0	0	0
Lupinus x regalis	2.2%	3	0	0	0	2	3	1.5 (±0.7)	0	0	0

***Table 8.2** continued...*

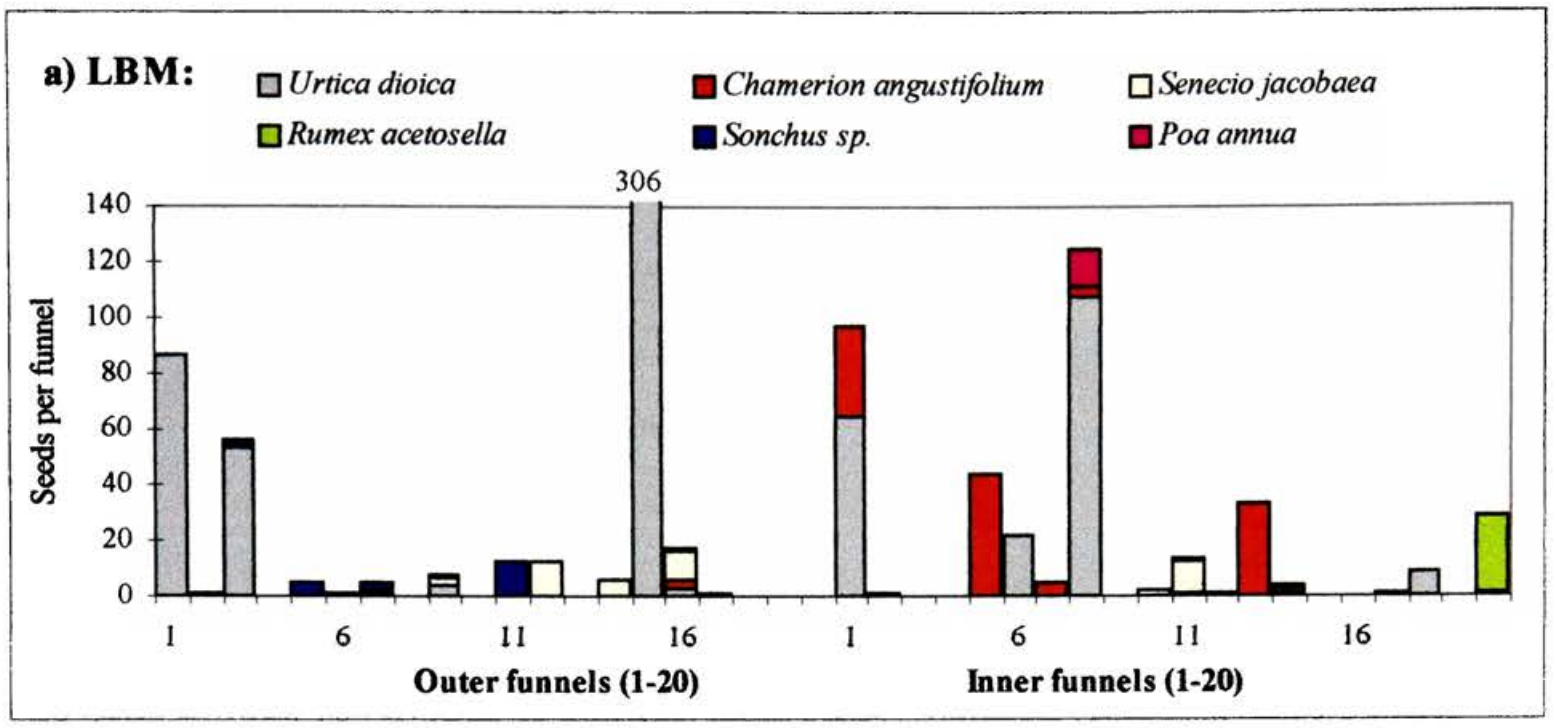
SPECIES	ALL SURVEY		LARGE BENTLEY MILL (LBM)			VINCENT DRIVE (VD)			WILSON ROAD (WR)		
	OCCURRENCE (% FUNNELS)	TOTAL SEEDS	SITE FUNNELS	SITE TOTAL	MEAN WHERE FOUND	SITE FUNNELS	SITE TOTAL	MEAN WHERE FOUND	SITE FUNNELS	SITE TOTAL	MEAN WHERE FOUND
Agrostis capillaris	2.2%	2	0	0	0	0	0	0	2	2	1.0 (±0.0)
Leucanthemum vulgare	2.2%	2	0	0	0	2	2	1.0 (±0.0)	0	0	0
Capsella bursa-pastoris	1.1%	1	0	0	0	0	0	0	1	1	1.0 (±0.0)
Chenopodium album	1.1%	1	0	0	0	1	1	1.0 (±0.0)	0	0	0
Cirsium arvense	1.1%	1	0	0	0	1	1	1.0 (±0.0)	0	0	0
Cirsium vulgare	1.1%	1	1	1	1.0 (±0.0)	0	0	0	0	0	0
Digitalis purpurea	1.1%	1	0	0	0	0	0	0	1	1	1.0 (±0.0)
Elytrigia repens	1.1%	1	0	0	0	0	0	0	1	1	1.0 (±0.0)
Plantago lanceolata	1.1%	1	0	0	0	0	0	0	1	1	1.0 (±0.0)
Rubus fruticosus	1.1%	1	0	0	0	1	1	1.0 (±0.0)	0	0	0
ALL SPECIES		2366		925	33.0 (±63.1)		731	23.6 (±46.2)	37		19.2 (±21.2)

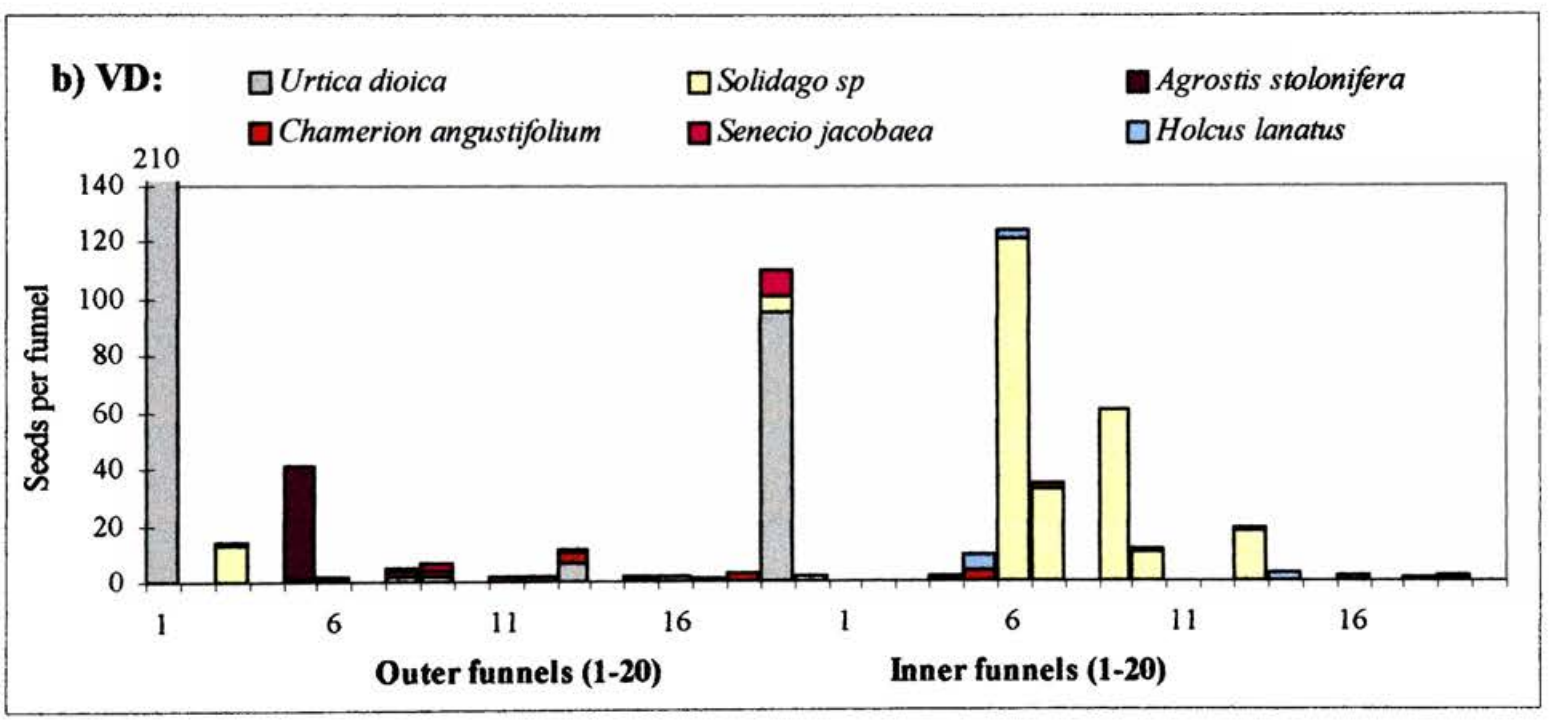
Total number of sites: 3, Total number of funnels recovered: 92, Total seeds found: 2366, Total number of species found: 36.

***Table 8.3:** Kolgomorov-Smirnov one-sample tests for the goodness of fit between observed patterns of seed deposition and the Poisson distribution (Two-tailed exact significance tests for divergence from expected distribution).*

	LARGE BENTLEY MILL (LBM)			VINCENT DRIVE (VD)			WILSON ROAD (WR)			ALL SITES
	All species	*Urtica dioica*	*Chamerion angustifolium*	All species	*Urtica dioica*	*Solidago* agg.	All species	*Solidago* agg.	*Holcus lanatus*	All species
n	28	28	28	31	31	31	36	36	36	95
Mean	33.0 ± (63.1)	23.5 ± (62.4)	4.5 ± (11.4)	23.6 ± (46.5)	10.2 ± (40.8)	8.7 ± (24.3)	19.7 ± (21.3)	6.8 ± (17.1)	5.2 ± (9.8)	24.9 ± (45.1)
Ks	3.58	4.16	3.66	4.08	5.03	3.58	2.65	4.66	3.51	6.24
Exact test (2-tailed)	p<0.001	p<0.001	p<0.001	p<0.001	p<0.001	p<0.001	p<0.001	p<0.001	p<0.001	p<0.001
Point probability	p<0.001	p<0.001	p<0.001	p<0.001	p<0.001	p<0.001	p<0.001	p<0.001	p<0.001	p<0.001

Figure 8.2: *Pattern of seed rain deposition for six most abundant species at each site*





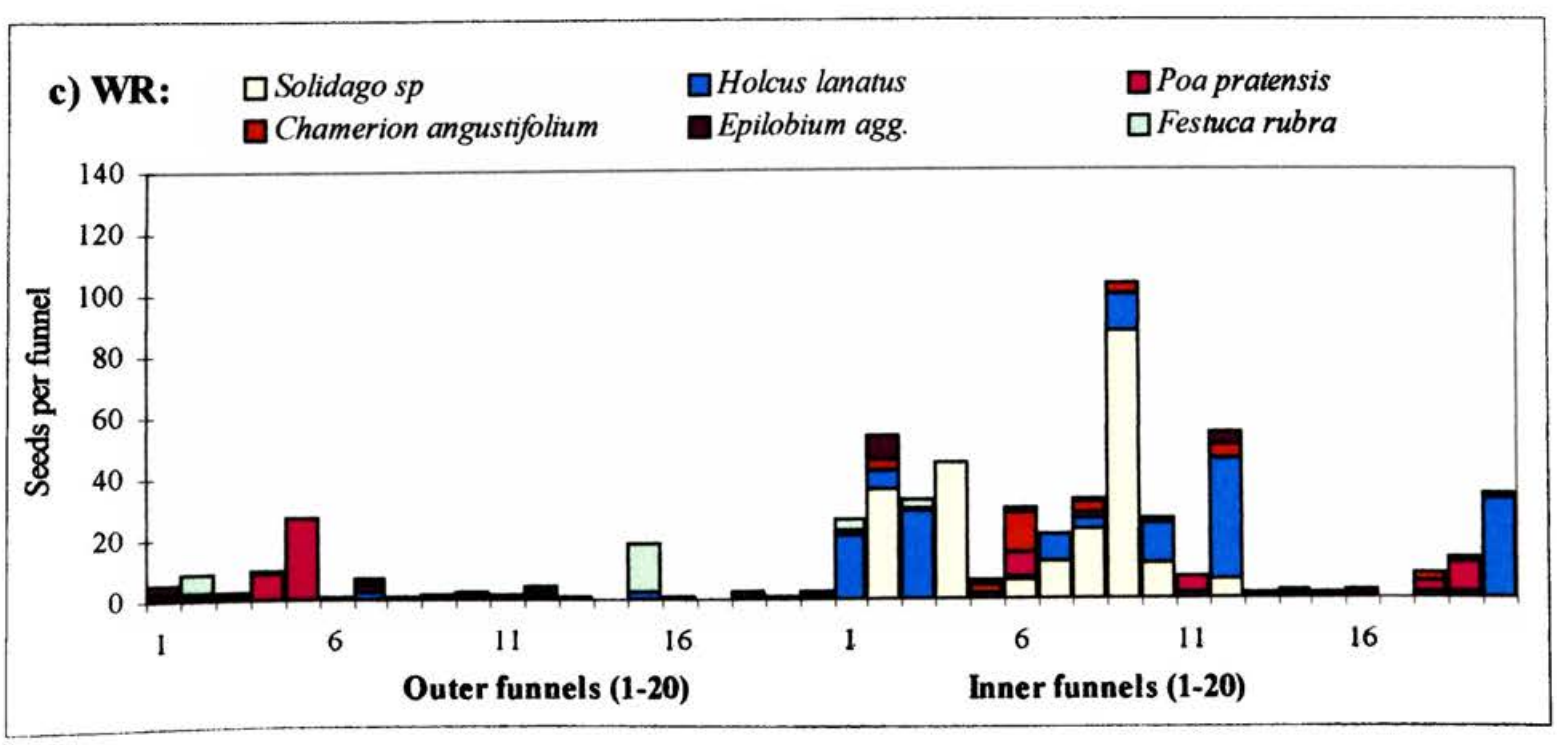


Figure 8.3: *Comparison between the species in the seed rain, ground vegetation and the seed bank.* ***(a-c):*** *The proportion of seed rain species that are also (i) in the top 20 species in the above-ground vegetation (ii) present but less abundant in the vegetation or (iii) not present in the vegetation at each site; similarly* ***(d-f)****: the number of seed rain species that are also (i) in the top-20 species in the seed bank, (ii) present but less abundant in the seed bank, or (iii) absent from the seed bank.*

Figure 8.4: *The height distribution of the top 20 most abundant species found in the seed rain, vegetation and seed bank at each site*

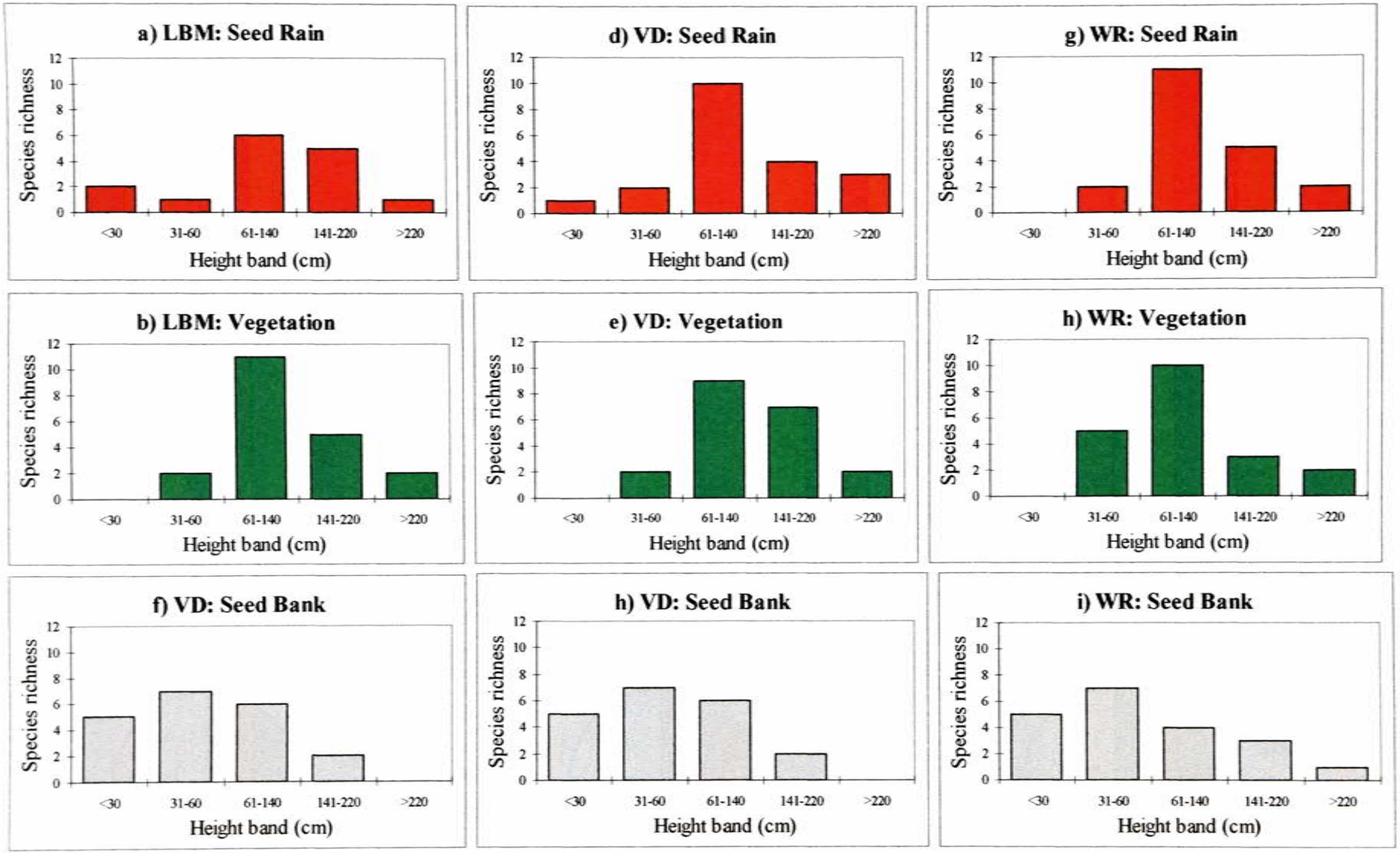


Figure 8.5: *Lifespan of top 20 most abundant species in the seed rain, vegetation and seed bank at each site.*

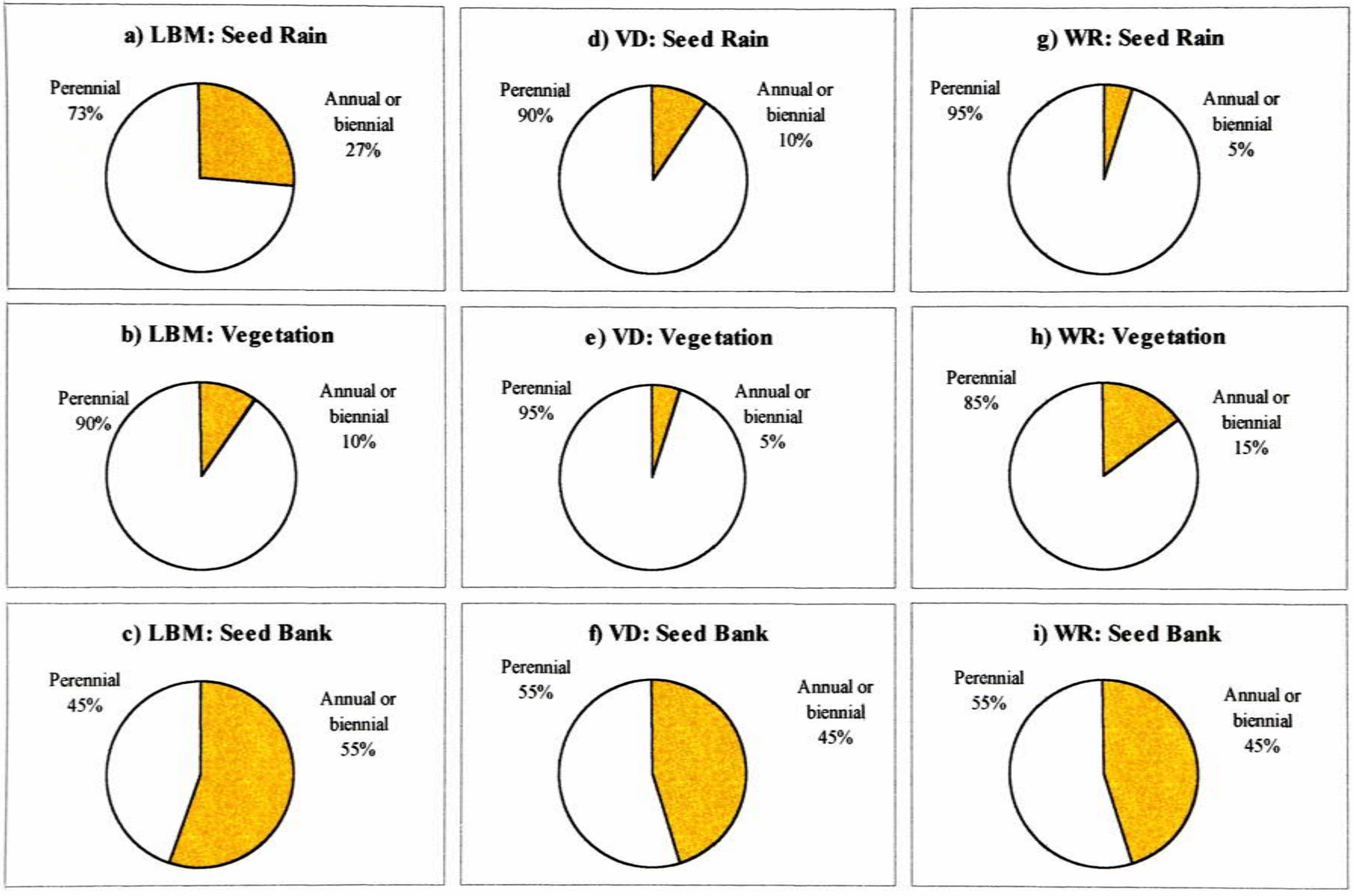
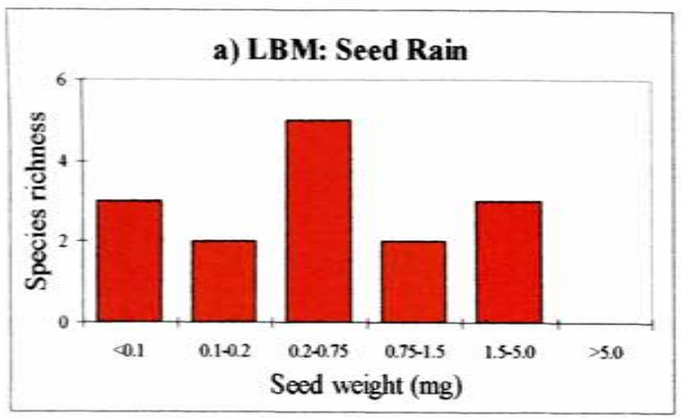
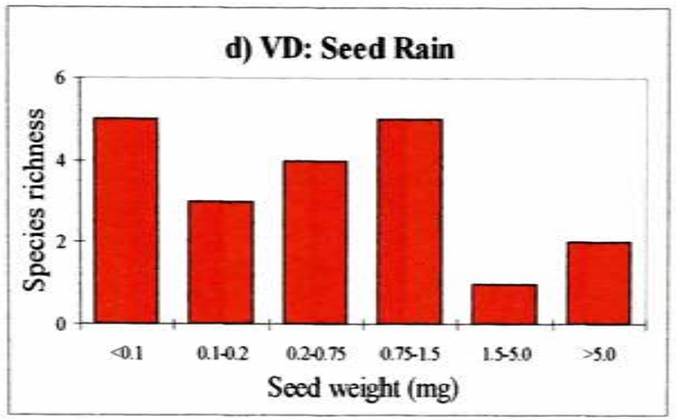
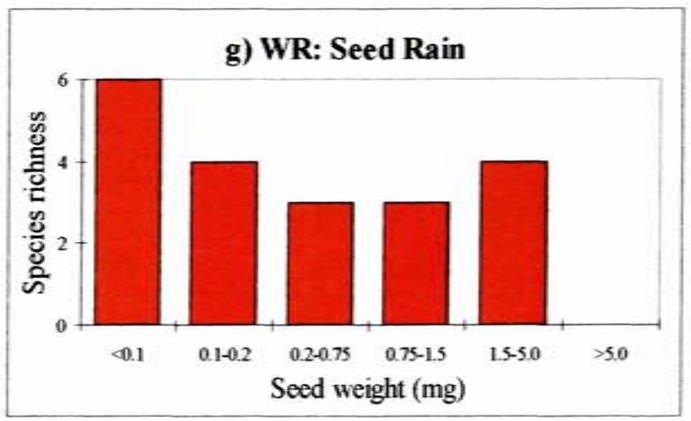
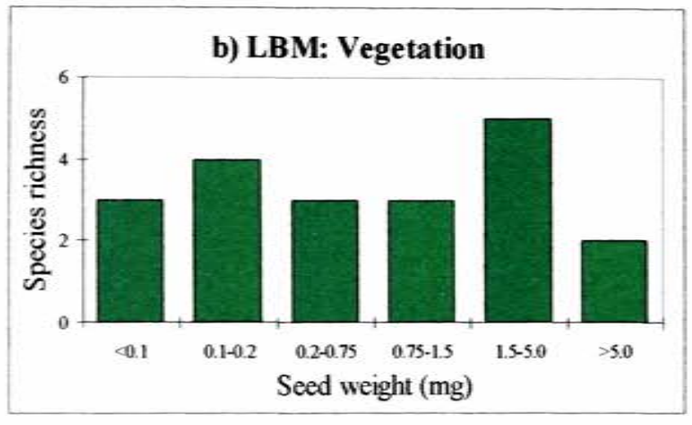


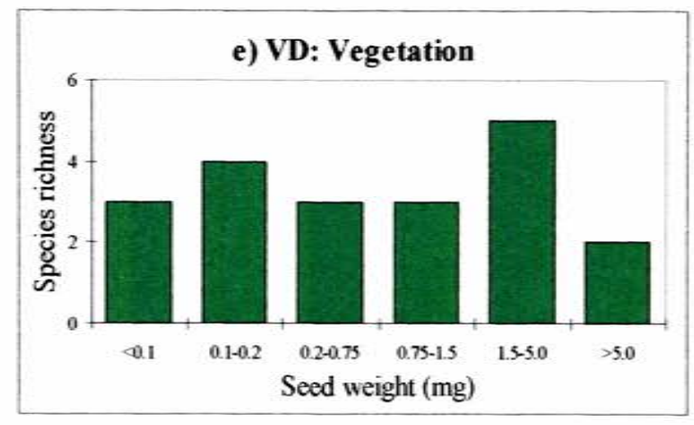
Figure 8.6: *Seed weight (mg) of top 20 most abundant species in the seed rain, vegetation and seed bank at each site.*

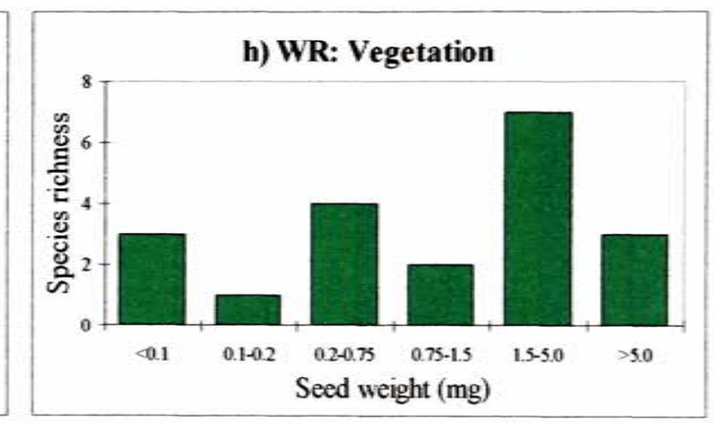


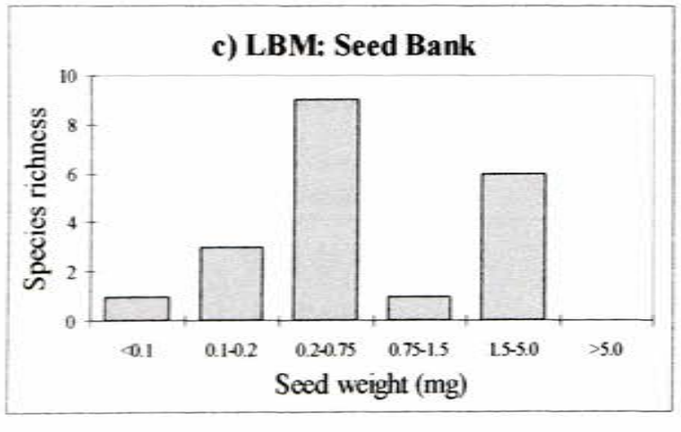


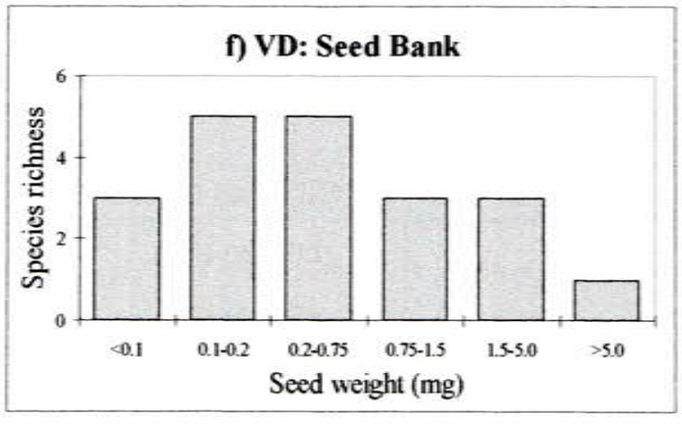












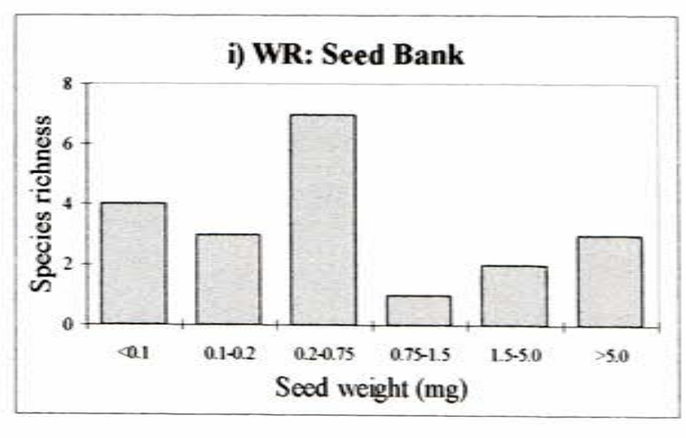


Table 8.4: *Differences in traits of the top 20 species found in (i) the seed rain, (ii) the vegetation and (iii) the seed bank at each site: (a) mean height (one-way ANOVA and Tukey Family Error tests); (b) representation of annuals and perennials (Fisher Exact Probability tests); (c) mean seed weight (one-way ANOVA and Tukey Family Error tests). †p<0.1, *p<0.05, **p<0.01, ***p<0.005*

(a) Mean height

	MEAN SPECIES HEIGHT (CM)			ANOVA	TUKEY FAMILY ERROR RATE		
SITE	Seed rain (SR)	Vegetation (V)	Seed bank (SB)		SR vs. V	SR vs. SB	V vs. SB
LBM	99.6 (± 51.1)	137.8 (± 104.1)	81.3 (± 60.9)	† F=2.71, p=0.08	n.s.	n.s.	†
VD	111.7 (± 60.8)	141.0 (± 99.4)	66.8 (± 40.7)	** F=5.45, p=0.007	n.s.	n.s.	***
WR	100.6 (± 39.0)	121.5 (± 107.6)	81.0 (± 63.2)	n.s.	n.s.	n.s.	n.s.

(b) Representation of annuals and perennials

	Seed rain (SR)		Vegetation (V)		Seed bank (SB)		FISHERS EXACT TESTS		
SITE	ANNUALS	PERENNIALS	ANNUALS	PERENNIALS	ANNUALS	PERENNIALS	SR vs. V	SR vs. SB	V vs. SB
LBM	4	11	2	18	11	9	n.s.	† p=0.091	*** p=0.003
VD	2	18	1	19	9	11	n.s.	* p=0.015	*** p=0.004
WR	1	19	3	17	9	11	n.s.	*** p=0.004	* p=0.041

(c) Mean seed weight

	MEAN SEED WEIGHT (MG)			ANOVA	TUKEY FAMILY ERROR RATE		
SITE	Seed rain (SR)	Vegetation (V)	Seed bank (SB)		SR vs. V	SR vs. SB	V vs. SB
LBM	0.68 (± 0.91)	1.5 (± 2.2)	0.76 (± 0.74)	n.s.	n.s.	n.s.	n.s.
VD	1.97 (± 4.8)	2.9 (± 5.9)	1.2 (± 2.7)	n.s.	n.s.	n.s.	n.s.
WR	0.83 (± 1.2)	3.9 (± 8.3)	2.7 (± 6.4)	n.s.	n.s.	n.s.	n.s.

Edge effects

Figure 8.7 shows the number of species in the vegetation that were unique to either the perimeter or interior areas of the sites, and whether they were also recorded in the seed rain there. In general, germinable seeds of the unique species were not captured in the funnels. This does not support the idea that a significant rain of seeds from an external source may be essential to their presence.

Table 8.5 shows the similarity between the seed rain species found at individual traps with both the immediate and next most adjacent (5m distance) vegetation quadrats. Similarity is scored here according to the percentage of species found in the seed rain at a funnel that were also found in the respective quadrats. Comparison between scores for traps of each sample area was made using paired t-tests. At all sample areas, similarity was highest to the immediate quadrat, suggesting species composition of the seed rain was determined largely by the composition of immediate vegetation. However, at all sites, the difference in similarity with immediate or adjacent quadrats was consistently higher at interior patches, suggesting that dispersal here is especially restricted.

Environmental measurements

Ellenberg values for the species lists generated for perimeter and interior patches were compared for differences in variance (Bartlett's Chi-squared test) and differences in mean values (two-sample T-Tests) (Table 8.6a). Interior and perimeter patches showed a similar degree of variance for the majority of values. Furthermore, the only instances in which greater heterogeneity was exhibited by the species values within a patch were for interior samples. The mean Ellenberg values exhibited were also broadly similar with the exception of nitrogen at site LBM, which pointed towards higher fertility at the edge. Mean Ellenberg values were also calculated for each quadrat (Table 8.6b) and tested as before to examine how patchy the environment may be within a site. The results were broadly similar to those described in Table 8.6a. The degree of variance exhibited by the perimeter and interior species lists only consistently differed for all sites for salinity. At all sites, greatest variation in salinity was shown between interior quadrats. Nitrogen values also showed greater heterogeneity at interior quadrats at site LBM. Only one test, for pH at VD, significantly demonstrated greatest

heterogeneity between perimeter quadrats. There were also few significant differences between mean quadrat Ellenberg values of perimeter and interior sample areas and, as with Table 8.6a, no consistent pattern emerged which could indicate an explanation for explain higher diversity at the edge.

Species exclusively found at the perimeter were compared with those found only at the interior and the shared pool of species for the representation of annual species and for mean seed longevity. Figures 8.8 and 8.9 and Table 8.7 demonstrate that the species which were unique to the perimeter communities in this study had significantly higher numbers of annuals and had seeds which remain viable in the soil for longer.

8.4 Discussion

The loss of materials is a predictable risk when carrying out research on urban sites which are used regularly by the general public. The methods employed for catching seeds had the benefits of being inexpensive, quick to set up and, if a longer period of study was being attempted, easily replaceable. Previous experimental researchers on seed rain have utilised a variety of sampling methods. Sticky traps (e.g. Wagner 1965; Werner 1975; Rabinowitz & Rapp 1980) can be prone to predation (Schott 1995). Removal of the bags in this study often revealed the presence of large numbers of invertebrates but few were seed-eating taxa and while it is impossible to estimate actual loss of seeds due to predation it is suggested that it was generally low. Traps designed to prevent seed predation have been designed. Mesh traps like those employed by Williams (1990) were designed for large seeded species in forest communities but are not applicable to the habitat type studied here. Schott's (1995) study utilised a funnel placed within PVC piping in the ground but with the funnel slightly raised to prevent predation. Placing the funnel as close to the ground as this may also be desirable to monitor precisely actual deposition but it was not possible to replicate this on the study sites here due to the rubble nature of the substrate. The shallow nature of the soil would have meant collection of water in a pipe setup and rotting of the muslin bags. Liability for accidents caused by placing ground level traps on sites used regularly by the general public was also an issue.

Figure 8.7: *The number of species in the vegetation that were unique either the outer or inner areas at each site, and whether they were also found in the seed rain*

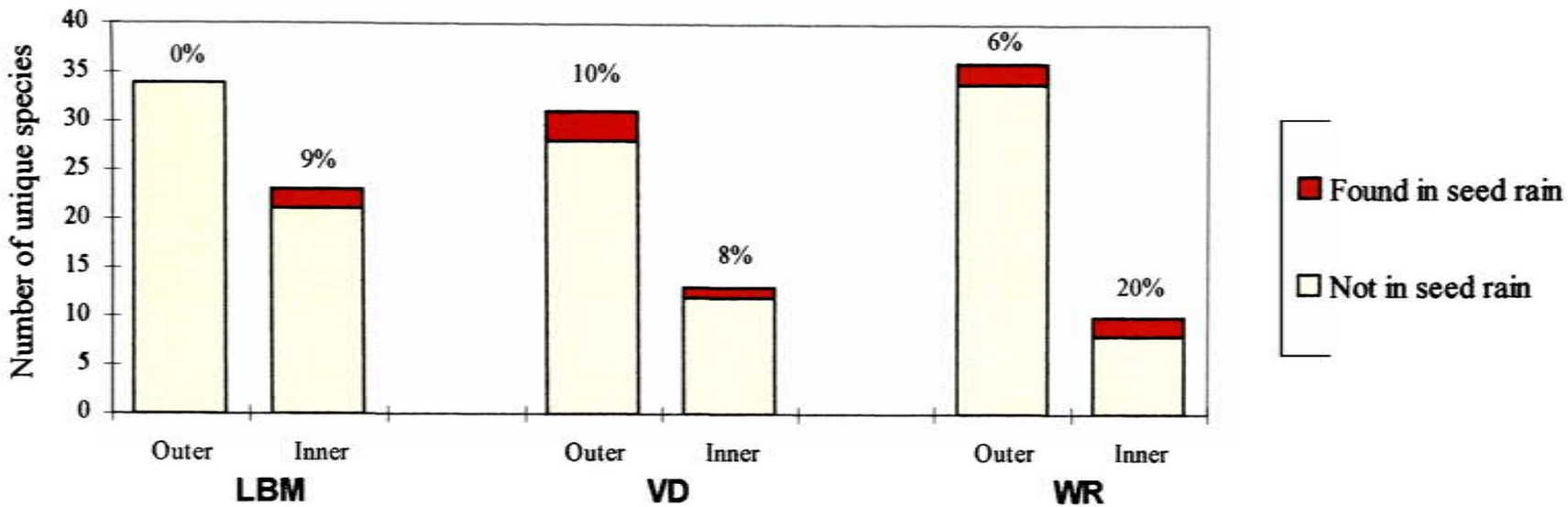


Table 8.5: *Paired t-test comparisons between the similarity of species found in each seed rain funnel & the vegetation in (i) the immediate quadrat and (ii) the adjacent quadrat at 5 metres distant.*

	OUTER QUADS			INNER QUADS			ALL SITE QUADS		
	Mean similarity to immediate vegetation	Mean similarity to vegetation 5m distant	t-test	Mean similarity to immediate vegetation	Mean similarity to vegetation 5m distant	t-test	Mean similarity to immediate vegetation	Mean similarity to vegetation 5m distant	t-test
LBM	0.48 ± (0.40)	0.38 ± (0.30)	n=13, T=0.59, p=0.280	0.62 ± (0.35)	0.32 ± (0.34)	n=15, T=2.5, p=0.013	0.55 ± (0.38)	0.35 ± (0.32)	n=28, T=2.13, p=0.021
VD	0.32 ± (0.23)	0.28 ± (0.19)	n=17, T=0.81, p=0.021	0.43 ± (0.33)	0.25 ± (0.33)	n=14, T=3.31, p=0.003	0.37 ± (0.28)	0.27 ± (0.26)	n=31, T=2.73, p=0.005
WR	0.46 ± (0.28)	0.31 ± (0.21)	n=18, T=2.41, p=0.001	0.60 ± (0.26)	0.38 ± (0.17)	n=18, T=3.08, p=0.003	0.53 ± (0.28)	0.35 ± (0.19)	n=36, T=3.92, p=<0.001

Table 8.6: *Comparisons of the mean and variance of species Ellenberg values between the perimeter and interior areas of sites using Bartlett's and Levene's tests for homogeneity of variance and two-sample t-tests.*

(a) Perimeter species list vs. interior species list

	SITE	MEAN OUTER	MEAN INNER	VARIANCE: Barlett's test	VARIANCE: Levene's test	MEANS: t-Test[1]
LIGHT	LBM	6.68 (± 0.70)	6.80 (± 0.71)	n.s.	n.s.	n.s.
	VD	6.95 (± 0.61)	6.89 (± 0.71)	n.s.	n.s.	n.s.
	WR	6.55 (± 0.90)	6.69 (± 0.75)	n.s.	n.s.	n.s.
MOISTURE	LBM	5.13 (± 0.74)	4.94 (± 0.96)	In>Out p=0.11	In>Out p=0.10	n.s.[2]
	VD	5.18 (± 0.85)	5.28 (± 0.74)	n.s.	n.s.	n.s.
	WR	5.35 (± 0.95)	5.47 (± 1.08)	n.s.	n.s.	n.s.
REACTION	LBM	6.45 (± 0.83)	6.53 (± 0.88)	n.s.	n.s.	n.s.
	VD	6.44 (± 0.85)	6.64 (± 0.84)	n.s.	n.s.	n.s.
	WR	6.28 (± 1.01)	6.50 (± 1.03)	n.s.	n.s.	n.s.
NITROGEN	LBM	6.18 (± 1.33)	5.53 (± 1.54)	n.s.	n.s.	* p=0.05
	VD	5.35 (± 1.27)	5.83 (± 1.52)	n.s.	n.s.	n.s.
	WR	5.63 (± 1.15)	5.78 (± 1.27)	n.s.	n.s.	n.s.
SALT	LBM	0.08 (± 0.27)	0.08 (± 0.28)	n.s.	n.s.	n.s.
	VD	0.154 (± 0.28)	0.37 (± 0.66)	In>Out p=0.04	n.s.	n.s.[2]
	WR	0.10 (± 0.30)	0.14 (± 0.42)	In>Out p=0.001	n.s.	n.s.[2]

(b) Mean perimeter quad values vs. mean interior quad values

	SITE	MEAN OUTER	MEAN INNER	VARIANCE: Barlett's test	VARIANCE: Levene's test	MEANS: t-Test[1]
LIGHT	LBM	6.73 (± 0.25)	6.81 (± 0.25)	n.s.	n.s.	n.s.[2]
	VD	6.89 (± 0.20)	6.97 (± 0.19)	n.s.	n.s.	n.s.
	WR	6.77 (± 0.21)	6.81 (± 0.23)	n.s.	n.s.	n.s.
MOISTURE	LBM	5.23 (± 0.30)	5.10 (± 0.37)	n.s.	n.s.	n.s.
	VD	5.28 (± 0.27)	5.34 (± 0.19)	n.s.	n.s.	n.s.
	WR	5.31 (± 0.28)	5.37 (± 0.33)	n.s.	n.s.	n.s.
REACTION	LBM	6.68 (± 0.28)	6.61 (± 0.35)	n.s.	n.s.	n.s.
	VD	6.42 (± 0.28)	6.55 (± 0.19)	Out>In p=0.11	n.s.	p=0.1[2]
	WR	6.31 (± 0.25)	6.53 (± 0.20)	n.s.	n.s.	*** p=0.004
NITROGEN	LBM	6.47 (± 0.57)	5.78 (± 0.95)	In>Out p=0.03	In>Out p=0.01	** p=0.009[2]
	VD	5.62 (± 0.55)	6.08 (± 0.50)	n.s.	n.s.	** p=0.009
	WR	5.51 (± 0.38)	5.66 (± 0.43)	n.s.	n.s.	n.s.
SALT	LBM	0.05 (± 0.07)	0.10 (± 0.12)	In>Out p=0.01	In>Out p=0.005	p=0.11[2]
	VD	0.18 (± 0.07)	0.16 (± 0.12)	In>Out p=0.04	In>Out p=0.03	n.s.[2]
	WR	0.10 (± 0.07)	0.17 (± 0.12)	In>Out p=0.03	In>Out p=0.05	* p=0.04[2]

[1] *t-Test assuming equal variance unless indicated.* [2] *t-Test assuming unequal variance.*

Figure 8.8: *The representation of annual and perennial species in the vegetation that were (i) unique to the perimeter; (ii) unique to the interior; and (iii) shared by both areas.*

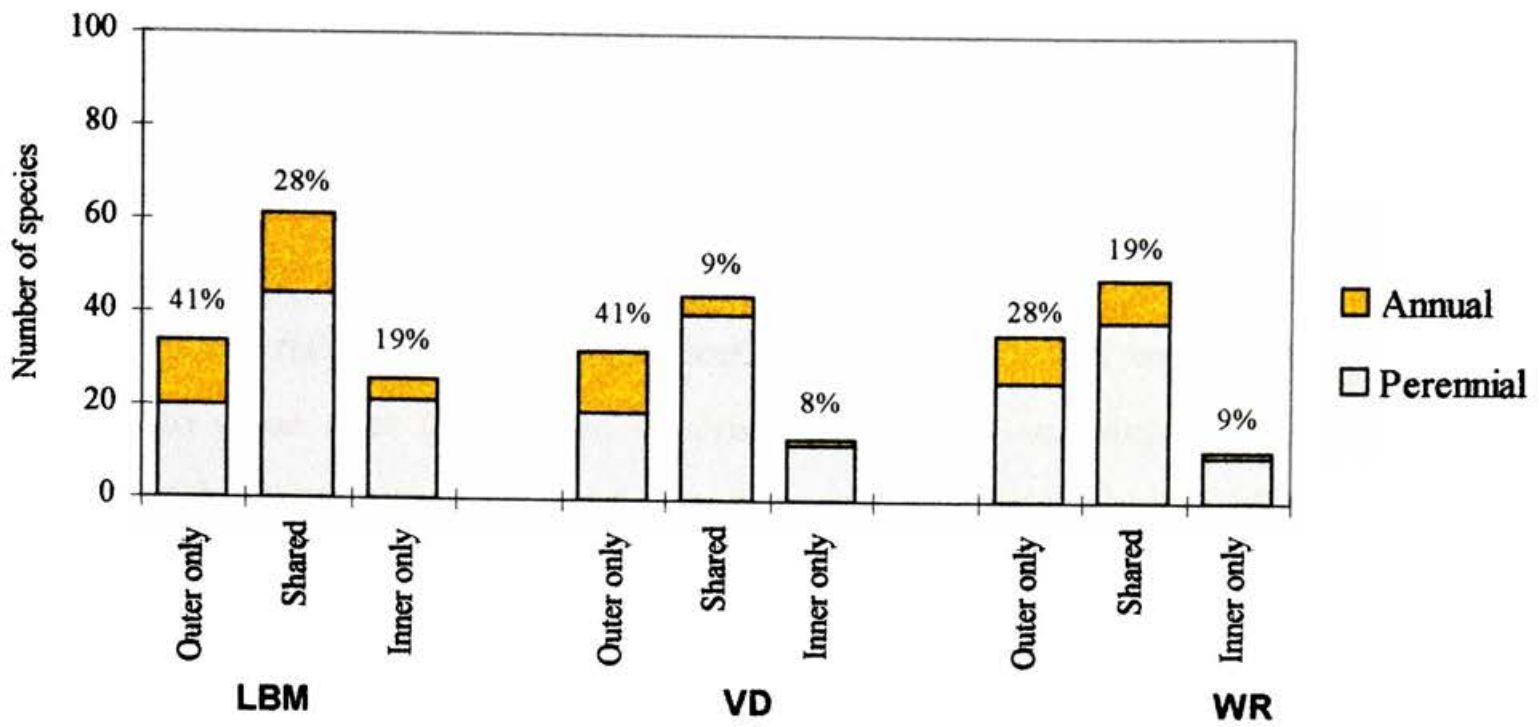


Figure 8.9: *Seed longevity of species in the vegetation that were (i) unique to the perimeter; (ii) unique to the interior; and (iii) shared by both areas.*

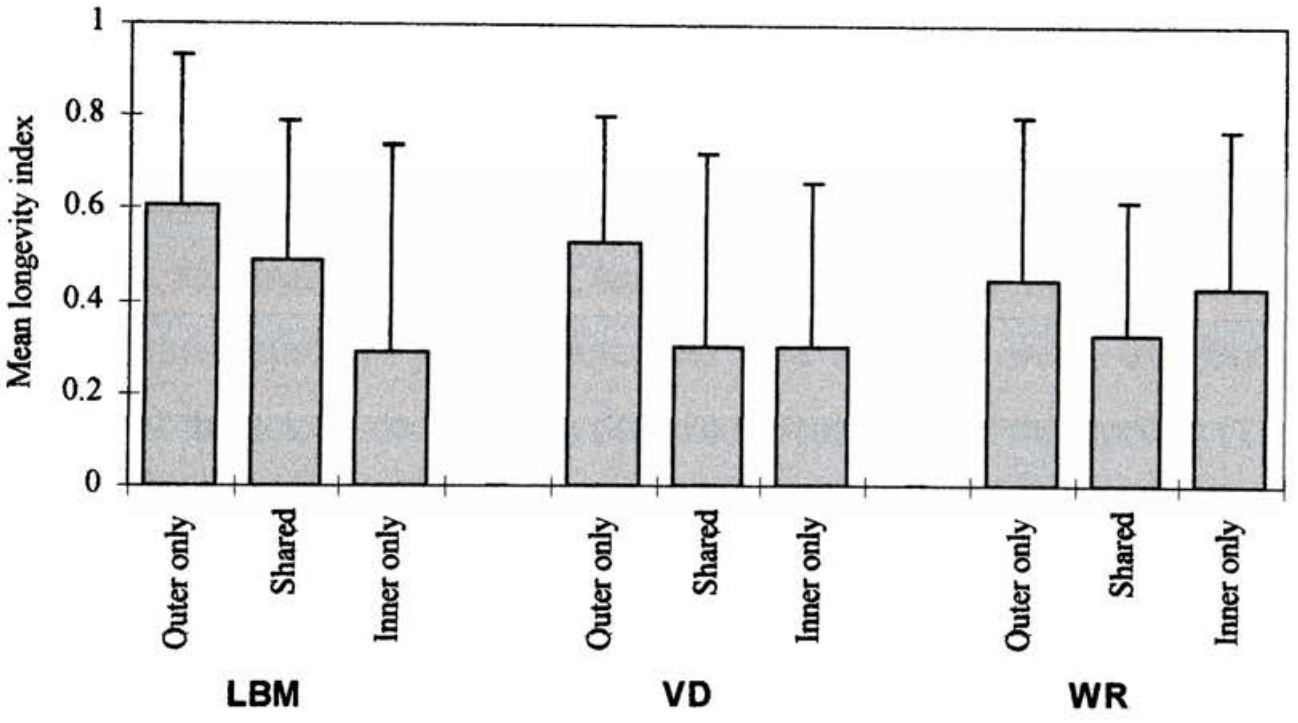


Table 8.7: *Differences in representation of annuals and mean seed longevity between species in the vegetation that were (i) unique to the perimeter; (ii) unique to the interior; and (iii) shared by both areas. (One-way ANOVA and Tukey Family Error Tests).*

	ANOVA	TUKEY FAMILY ERROR TESTS		
		Outer vs. Inner	Outer vs. Shared	Inner vs. Shared
%ANNUALS	F=8.13, p=0.02	p<0.05	p=0.05	n.s.
LONGEVITY	F=11.73, p<0.01	p<0.05	p<0.01	n.s.

As with studies of soil seed banks, the use of emergence methods alone cannot be expected to provide a complete picture of the seeds present. The transient nature of many species' seeds and the need for scarification of seed coat or specific temperature requirements for germination will lead to some species present in the bags being either under-recorded or not recorded at all. Nevertheless, it is widely accepted for studies where a high sampling intensity is required that emergence methods are the most practical approach. It is however likely that the major factor reducing the recorded seed rain in the funnels was the tendency of the vegetation to close over them often covering many traps completely. The survey period coincided with the maximum seasonal growth period. The patchy nature of the trap contents highlights very clearly the observation that any attempt to gain an overall picture of deposition at a site requires a high density of traps (Kollman & Goetze 1998) and consequently the approach used here can be recommended for its time efficiency and cost. The differences in traits representation between vegetation, the seed rain and the seed bank illustrate the difference between the majority of seeds being deposited and those which actually form a persistent seed bank. The seed rain is dominated especially by the tall perennial species which also dominate the vegetation, while the species found in the soil at the sites studied were predominantly short annual species typical of earlier successional stages. The fact that the seeds of *Chamerion angustifolium* were the most well distributed outside of the vicinity of parent plants and were the most evenly distributed generally is a reflection of the acknowledged high dispersal capabilities of this species (Thompson, Gaston & Band 1999).

The close relationship between the immediate vegetation and the seeds trapped and the poor representation of perimeter-only species in the rain is not surprising considering the observations that the vast majority of seed set from most species studied falls to rest within a few metres of the parent plant. This can of course be expected to be highly variable, depending on taxa and their respective life form, mechanisms of seed release, mode of dispersal and seed morphology. However, taken as a whole, the results suggest it is unlikely that many species can occur as sink populations in the manner described by the mass effects hypothesis. The limited dispersal of most species over the long to medium range is only likely to be able to serve the purpose of enabling initial colonisation on a favourable site, and repeated recolonisations from outside sources are unlikely. This is likely to be particularly the

case as vegetation develops and potential gaps become infrequent.

There is no evidence to suggest that any of the environmental variables tested using Ellenberg values contribute significantly to differences in species richness between edges and the centre at the derelict sites studied here. These potentially important factors were apparently very similar across all sites and there was no evidence for greater stress e.g. through less water availability or low fertility, at the edges. Furthermore, the similar amount of variance exhibited for these values at perimeter and interior sites does not support a model of resource complementarity. This lack of variance also provides further evidence against a mass effects hypothesis as species existing as sinks may have been expected to have had different environmental values to species forming self-sufficient populations, particularly where the boundary habitat is very different.

However, other environmental components not considered by utilising Ellenberg factors may be especially important in such highly anthropogenic landscapes as cities. This may be particularly the case for sites situated adjacent to roads. Pollution caused by the emissions of motor vehicles may alter the composition of the vegetation at site edges adjacent to roads. Elevated levels of CO_2, NO_2 and SO_2 are all commonly recorded beside roads (Gilbert 1989). The zone of influence on the plant community is influenced by the density of traffic on the road (Angold 1997) and so is likely to be highest in urban areas. The concentration of high numbers of annuals at the edges could be accounted for by pollution. Both Hunt *et al.* (1991) and Bazzaz & Garbutt (1988) found that higher levels of CO_2 prevented late successional species achieving complete dominance over early successional species, so that a more open vegetation situation remains in which annual species can persist.

Nevertheless, the high proportion of annual and persistent seeded species found amongst the species unique to edges was consistent across all sites despite Site VD being a distance of over 400 metres from the nearest road and Site WR being adjacent to a quiet supply road. Consequently, the results lend greater support to disturbance being the mechanism which ensures higher diversity at derelict edges. A high intensity of illegal tipping of building and garden waste tends to occur at the very edges of sites and they are also more likely to be subject to heavy trampling than the interiors. These activities are likely to prevent, reverse or reduce the extent of competitive exclusion. The results of this study suggest that the principal

benefactors are those species present in the soil seed bank which are representative of an earlier successional phase in a site's history but which have been suppressed over time in the interior of a site. The ability of these species to reproduce quickly and produce high seed numbers may enable them to proliferate after disturbances, at least in the short term, but in the long term they will be dependent on the regularity of these events, if they are to persist in the vegetation. All of the sites in this study were mid-successional. Further research to test this hypothesis should concentrate on how the extent of any edge effect varies between sites of different ages and known disturbance history. The effect would be expected to be more pronounced on older sites where the interiors have remained undisturbed for longer, but where external activity remains high at the edge.

While the evidence of this research strongly suggests that disturbance plays the primary role in determining species richness, it is possible that the consequential wider availability of gaps may mean a secondary role for the available source pool of species. This may potentially be through the seed rain, as even infrequent seed deposition from outside could enable constant recolonisations of some species if enough gaps are generated but may be perhaps more significantly from anthropogenic sources. Arrival of propagules to site edges by other means than the natural rain is especially high beside roads. Mud on vehicles and bicycles may contain considerable numbers of seeds of numerous species (Clifford 1959; Scott & Davison 1985; Lonsdale & Lane 1994; Hodkinson & Thompson 1997), although those with small seeds are most likely to benefit from this mode of dispersal (Scott and Davison 1985; Hodkinson & Thompson 1997). In addition, garden waste is deposited predominantly on the edges of sites adjacent to roads and may account for increased numbers of annual weeds found in these situations and can often also lead to the short term establishment of cultivated species. Hill *et al.* (1994) regard the potential significance of anthropogenic dispersal as very high especially for alien species.

8.5 Conclusions

a) Higher species richness was observed at the edges of all three derelict sites studied. This could be mostly attributed to increased numbers of annual and/or persistent seeded-species.

b) The seed rain trapped at any particular location on the derelict sites studied was dominated by species found in the immediate vegetation and showed poor similarity to seed bank below. There was little evidence to suggest that the seed rain contained seeds from extraneous sources which could enable the persistence of sink populations.

c) Environmental differences between perimeter and interior sites were found to be low when species Ellenberg values were compared. Consequently using these variables neither greater environmental stress nor complementarity of resources could explain higher species richness at the edge.

d) a higher intensity or frequency of disturbance at the site perimeter enabling the regeneration of annual species typical of earlier successional stages was considered to be the primary factor which led to higher observed species richness in edge locations. Regeneration may occur from the seed bank, occasional propagules in the seed rain from outside sources or through other means of arrival particularly anthropogenic dispersal.

e) An additional possibility which requires further research is that pollution may have a role to play in damping down competitive exclusion at site edges in cities, particular beside roads, enabling a higher diversity of species especially annuals.

CHAPTER NINE

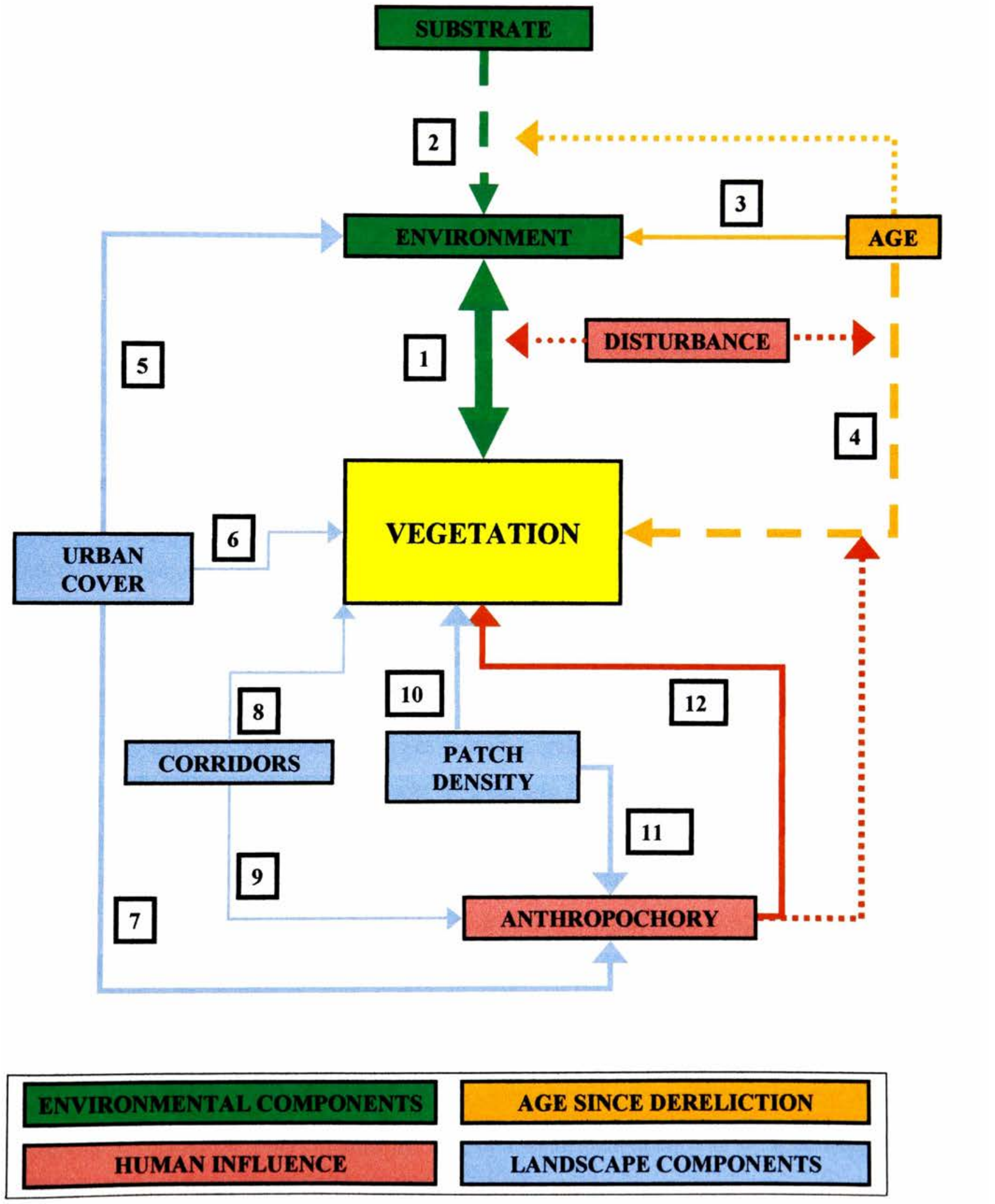
CONCLUSIONS

9.1 WHAT FACTORS DETERMINE THE SPECIES FOUND ON URBAN DERELICT SITES? A REVIEW OF THE EVIDENCE FROM A COMPREHENSIVE SURVEY OF THE DERELICT FLORA OF THE WEST MIDLANDS CONURBATION.

The findings of the research contained within this thesis have made it possible to put forward a considered opinion as to what are the major factors which influence the species composition of derelict sites in the West Midlands. The flow chart in Figure 9.1 illustrates the interaction of the major factors tested in the thesis and makes assertions as to their relative significance. Each relationship is denoted by an arrow of different weight based on how significantly it has been demonstrated (or inferred) in the research contained within the preceding chapters. Each relationship is also identified by a numbered box and are summarised below:

1 The **environment** has been demonstrated to be by far the most significant factor tested for its influence on the **species composition** of a site (Tables 6.3-6.5). The changes in species composition and their characteristic traits that occur during succession on derelict sites was demonstrated to be correlated with changes in the site environment (Tables 3.5-3.6) The most clear changes were a decline in both fertility and pH. In addition the different functional groups of species identified on later successional sites were shown to be adapted to different ranges of environmental preferences (Figure. 3.7). Part of the change in environment that occurs during the process of succession is related to feedback from the vegetation that is present, the build up of leaf litter, increasing shade and withdrawal of nutrients from the soil. Disturbance events such as isolated fires and tipping occur with very variable frequency across the urban zone and may often supplement the diversity of environmental niches available. It is these stochastic events which lead many sites to have very heterogeneous vegetation and high overall species richness. At the more extreme scale, site disturbances, where the whole site is burnt or subjected to intensive cutting or mowing, can create a more homogeneous environment and consequently can lead to a reduction in species diversity. The

Figure 9.1: *Flow diagram of major interactions determining the composition of urban derelict flora. Solid arrow denotes affect of one factor on another. Dashed arrow denotes effect significantly controlled by another factor Dotted arrow denotes controlling factor. Weight of arrows equals the postulated relative significance of a relationship.*



pattern essentially follows the intermediate disturbance models for maintenance of species diversity put forward by Odum (1963), Grime (1973, 1979) and Connell (1978, 1979).

2 & 3 The age of a site, which here has been treated as the time since it fell derelict or last underwent a major denudation, is also an important consideration when attempting to predict the assemblage of species which might be found. An older site will have a different environment for several reasons. One of these is the feedback already discussed from the vegetation that grows on it during the different successional stages. However, the older a site is the longer it will have been exposed to climatic factors, particularly rainfall, which may leach minerals and nutrients from the soil altering potentially both its fertility and pH. Without controlled experiments, in which vegetation is not allowed to establish it is not possible to predict accurately how significant the climatic aspect of site ageing is to the environment but it has been postulated here to be of moderate importance. The other important feature of an older site is that it has had more time to acquire potential immigrant species from the surrounding recruitment pool. However there was no evidence to suggest that it was predominantly heavier-seeded species which took longer to arrive at a site (Table 3.2). It seemed that species most likely to be present soon after the onset of succession were ruderal, therophytes, with high seed longevity, suggesting that presence as dormant seeds on denuded land may be the most important characteristic for initial colonisation success (Tables 3.2 & 6.6, Figures 3.5-3.6). However the seed bank at most sites was dominated by only a few species (Figure 4.4) and apparently cannot explain the diversity of species found at early successional sites. It seems that human activity again probably plays an important role influencing the relationship between age of sites and the number of species recruited. The tipping of garden waste, which occurs at the edges of some sites, offers the potential for many heavy-seeded species to arrive earlier than might be expected if only natural dispersal methods were occurring. This explains the lack of a significant seed weight-succession relationship. Furthermore, high seed longevity is also advantageous for being present in the soil of dumped pots and garden waste. Fire, quad biking and other activities, which disturb the existing vegetation, also break up and distort the natural relationship between age and succession, enabling the regeneration of species from the seed bank formerly suppressed by competitive vegetation. This explains the lack of a significant decline in the seed bank density with age (Figure 4.9).

4 **Substrate** was shown to be very significant in determining the stands of plant communities found on a site (Figure 2.7) and the overall assemblage (Tables 6.3-6.5). This is because it is an important component factor of the **environment** of the site demonstrated in Table 6.3. Brick rubble is particularly basic due to the lime mortar contained within it. Ballast and sandy soils tended to be more free draining providing a more xerophytic environment. Agricultural and old garden topsoils are typically highly fertile. These differences in environment due to the underlying substrate were also shown to have an effect on the speed at which succession progressed (Figure 2.7). The significance of the effect of substrate on the environment and ultimately the vegetation of a site was also however highly variable, depending on the age since dereliction. As sites grew older and succession progressed, the differentiation, which was quite clear on young sites of different substrates, gradually became less clear, as convergence towards very similar mesotrophic grassland and woodland understorey communities occurred.

5, 6 & 7 The **amount of urban cover** surrounding a site showed little significant direct relationship with the environment and nor with the vegetation taken as a whole (Tables 6.3-6.6). The two agricultural urban edge sites were very distinct but this seems to have had more to do with the previous land use and substrate than with actual location. There are three related reasons that may explain these findings. The first reason could be that the amount of disturbance and development that has occurred in lowland UK is so great that there is little significant difference in the conditions that sites in cities and rural areas are subjected to. There is however sufficient evidence in the literature regarding changes in the environment along urban-rural gradients to refute this supposition. Increased pollution (e.g. Gilbert 1971, Seaward 1976), highly modified soils (Rebele 1984,1992) and higher night temperatures (Chandler 1965) are typically recorded towards the city centre and may all have an impact on plant growth rates and the outcome of interspecific competition. The other remaining possible explanations relate firstly to the location of the sites and secondly to the nature of the habitat studied. All the sites studied here were within or at the edge of the West Midlands conurbation and consequently by definition were reasonably urban. No site had no urban cover within a 1km radius. It may be that without surveying truly rural derelict locations we not enough of a gradient was sampled to have observed an urban-rural effect. This could be applicable both in terms of an environment gradient and in terms of a recruitment pool

gradient. The final reason may simply be that all derelict sites are by definition highly hemerobic (in the sense of Kowarik 1992), as they have high anthropogenic involvement in their origin. Such artificial sites are much less likely perhaps to demonstrate a significant response to urbanisation than natural or semi-natural habitat patches, such as woodland, which may be clearly species-depauperate in cities (Goldsmith 1988), because the species within them are better pre-adapted to survive under anthropogenic pressures. However, while urban cover did not explain a significant proportion of the variation between the total species assemblages observed on sites, it did account for differences between the size and nature of their alien flora. The proportion of alien species was significantly higher on more urban sites (Table 7.6). This was due to much higher proportions of neophyte aliens, while the actual proportion of archaeophyte aliens was not significantly affected (Table 7.5). The results apparently indicate the changing nature of species immigration to this country. Archaeophytes pre-1500 AD were largely introduced accidentally as grain weeds and are consequently associated most strongly with rural areas, while neophytes post 1500 AD are more likely to have come over deliberately as garden plants and are consequently most associated with cities. Despite this finding the most frequent aliens as a whole across the city were mostly archaeophytes (Table 7.1) indicating the importance of duration of presence here to current status. The flow diagram also suggests that urban cover probably has a relationship with the degree of anthropochory that occurs. This is simply inferred here on the basis that higher population density and traffic movement associated with more urban cover offers more opportunity for accidental or deliberate dispersal by humans.

8 & 9 The evidence from this study suggests that the significance of linear features, **urban corridors,** is low. Sites show no greater similarity to each other if adjacent or close to these interconnected features (Tables 6.3-6.6) suggesting that the corridors do not increase connectivity to any great extent. A small number of species do however show a significant positive association with sites adjacent to corridors (Tables 6.7-6.8). These are however predominantly species which are well distributed on the two corridor habitats studied themselves suggesting that the proximity of source patch may be important rather than that corridors enhance dispersal potential. However amongst the species associated with both rail and river corridors the proportion of alien species is significantly higher than expected (Chapter 6 p.165).

10 & 11 Similarity of sites within **high-density derelict areas** is higher suggesting greater inter site dispersal of propagules. Of the species which were particular typical of sites in these high density regions (Table 6.9) a significantly larger than expected proportion were members of the Fabaceae, which typically are heavy seeded and not wind dispersed and consequently perhaps amongst the potentially more vulnerable taxa to isolation due to fragmented habitat. However whether the dispersal of seeds of these species is predominantly natural, which for legumes is typically by being consumed by bird or animal and passing through the digestive tract, or by anthropochory is unclear. Greater proximity of sites increases the likelihood of both natural forms of dispersal but it is also the case that high density areas of derelict land typically have a correlated high level of human activity in the West Midlands as they are the principal areas of redevelopment. In particular the movement between these sites of bulldozers and other construction vehicles may represent a strong likelihood that transfer of heavy seeded propagules between these sites will be higher.

9.2 A REVIEW OF THE THESIS OBJECTIVES: THE CONTRIBUTION OF THIS STUDY TO UNDERSTANDING IN THE FIELD OF URBAN ECOLOGY

The introduction for this thesis set out four principle objectives, which the research methods employed have set out to fulfil. Below is a critical review of the extent to which this work has fulfilled these objectives. This includes a summary of the key contribution of the work to current understanding in urban ecology, a recognition of where the research has not fully been able to address the questions put forward, and an identification of further research needs to develop further the areas covered.

Objective 1: Investigate the diversity of plant communities which occur on derelict land and examine how well these can be attributed to existing vegetation associations described in the National Vegetation Classification (Rodwell 1992-99) and by Shepherd (1996).

The study attempted to sample as wide a variety of derelict sites representative of the West Midlands conurbation and by covering fifty sites represented one of the most significant surveys of this habitat that has taken place in the United Kingdom. Nevertheless some consideration should be given to what types of dereliction were not investigated, as the

findings of this study may not necessarily be applicable across the broader spectrum which occurs nationally. It would be fair to say that the majority of sites visited for this work could be described as 'soft' derelictions because contamination and the hostility of the substrate to recolonisation of vegetation was typically low. The speed of regeneration and the most successful plant strategies applied on the West Midlands sites may be expected to differ to some extent from those which occur on 'hard' substrates such as colliery spoil, metal mines, salt beds or over highly toxic sites formerly occupied by chemical plants. When considering the processes that take place on these habitats, researchers would be recommended to refer to specific research in these areas (e.g. Kent 1982, 1987; Prach 1987; Cohn *et al.* 2001). However it can be expected that the diversity of pioneer communities found over these types of derelict habitats is also considerable, as demonstrated quite clearly for the West Midlands survey here. This only serves to reinforce the conclusion reached from the findings of this thesis that the application of a vegetation classification approach to early successional vegetation on these habitats is not likely to be of great use in understanding the processes determining species composition on individual sites.

Utilising the classification approach across such a large number of sites of different ages did however provide a significant finding from the perspective of maintaining diversity in urban areas. Convergence of communities was clearly demonstrated over the chronosequence on the majority of common substrates, e.g. brick rubble, topsoils, refuse infill. Furthermore these mid to later successional communities were typically of much lower interest botanically, typically approaching species poor mesotrophic grassland and scrub communities, compared to some of the genuinely interesting and diverse pioneer communities. Consequently species and community richness on derelict sites is dependent either on continued turnover of sites by demolition and redevelopment and/or by direct intervention on sites to prevent succession proceeding or at least maintain small-scale site heterogeneity.

The huge pressure for development of brownfield sites to provide new houses and service sector industries makes it seem inevitable that there will be a reduction in extant sites, and that new sites may be developed so rapidly that no significant vegetative colonisation can occur. It may also be viewed unlikely that local government would commit money to 'manage' these sites in such a destructive way to maintain species richness, particularly as derelict sites are not seen as favourable surroundings for the majority of city dwellers. As

such there will in the probably be a considerable fall in open derelict habitats in the next five to ten years and concomitant loss of species diversity at the local level.

Some mitigation against this may come from the greater awareness that city naturalists now have of the potential value of these sites. The Local Biodiversity Action Plan process has broadened this awareness further, to the extent that, the majority of plans that cover major urban areas include urban wastelands as priority habitats. Specific targets are set for LBAP habitats, and councils are supposed to be committed to achieving them. As such objections raised to development of the most valuable sites will have added weight and planners will have to bring them into their considerations.

Objective 2: Describe the composition of the soil seed bank on urban derelict sites and compare the findings to expectations derived from studies of other habitats, in particular, with regard to the functional attributes of the component species. The wider aim was to identify the extent to which recruitment from the seed bank may play a role in maintaining biodiversity on these sites.

The seed bank investigation, carried out as part of this thesis, represented one of the most intensive studies of this kind ever undertaken. Over 36,000 seedlings were germinated and identified in a period of four months growing time. This was only possible due to the adoption of methods designed to condense large amounts of soil (Ter Heerdt *et al.* 1996) and the study provides a strong recommendation for this approach in future seed bank research. In addition, the work comprised the first significant examination of seed banks on urban derelict sites and as such the data set alone provides a significant contribution to current knowledge in this field. Furthermore, the study provided good comparison with other work on disturbed and successional habitats.

Although it should again be noted that these were predominantly 'soft' derelict sites and do not represent the full diversity of these habitats in the UK, it was noticeable how similar the seed banks of different sites were to each other. In particular, a few consistently occurring species dominated the seed banks at the majority of sites. This was demonstrated to occur even where the vegetation above differed considerably. Less surprisingly, the seed banks were shown to be made up of a significantly higher number of annual species than the vegetation. This is to be partly expected, as plants adopting the annual life-form strategy are of course

dependent on regular regeneration, from seed. However it may not have been expected to be so clear a finding as the vegetation on derelict sites had a much higher proportion of annuals than more stable habitats anyway.

Seed density was found to be particularly high in general, as expected for unstable habitats. Seed density declined with increasing soil depth (inferring earlier deposition) within each site, suggesting that a considerable number of species are only able to regenerate from the seed bank for a short period of time, after their disappearance from the vegetation. This is due to the absence of a long-term persistence strategy. The existing seed longevity index for species listed in Thompson, Bakker & Bekker (1998) providing the best means of explaining depth distribution according to functional attributes. Seed mass also proved a good predictive measure, lighter seeds as expected found in greater relative proportions at depth. However interestingly the combination of shape with mass, as put forward by Thompson, Band & Hodgson (1993) did not improve the ability to predict depth distribution. The significance of disturbance on anthropogenic sites, at breaking the link between shape and burial may be high, however other recent studies (Funes *et al.* 1999; Moles, Hodson & Webb 2000) have also stressed a lower importance of seed shape in determining depth and persistence.

Several changes in the composition of the seed bank were also observed along the site chronosequence. A significant decrease in similarity between seed bank and vegetation was observed along the chronosequence, and perennial species became more significant components of the seed bank. Similar findings have been observed on other successional seres (e.g. Donelan & Thompson 1980; Roberts & Vankat 1991). Nevertheless the relative abundance of annual seeds did not decline significantly, as may be expected along an undisturbed sere. This finding again points strongly to the probable influence of continuing small scale anthropogenic disturbances, particularly fire and fly tipping, as a contributing factor to determining the make up of the seed bank and the vegetation above. The haphazard frequency and timing of these disturbance events and the renowned spatial heterogeneity of seed banks within sites (Thompson 1986), from which regeneration occurs, are probably the most significant factors determining the highly patchy vegetation mosaic which is typical of many urban derelict sites. Indeed the seed rain investigation carried out within this thesis completed the observation of this regeneration cycle and also clearly demonstrated that seed deposition occurs in a strongly clustered fashion.

Objective 3: Identify the extent to which the functional attributes exhibited by species on these sites may be related to successional age and relate the findings to the causal factors which select for different plant strategies on derelict land.

The use of functional characters to explain patterns of species occurrence in this research was a central part of the approach used within this thesis to develop a wider understanding of the ecological processes dominating derelict habitats. Indeed when considering the relative distributions of nearly 400 species it is simply a matter of practicality to condense the information into comparable consistent traits. Existing knowledge about the relationship of several different plant characters and strategies with environmental constraints is such that most of the individual hypotheses incorporated within each research chapter of this work could be tested, to some extent, by examination of differences in trait representation. Not surprisingly therefore, functional evidence forms a considerable component of the results exhibited throughout the thesis, and contributed significantly also to achieving the goals set out in *Objectives 2 & 4*. The work laid out in Chapter 3 which considers the functional changes along the site chronosequence was however unique to the thesis, being the only section of work where the analysis and research were solely based on a functional approach.

That research set out to compare changes in plant trait representation along the chronosequence with those observed by other authors (e.g. Glenn-Lewin *et al.* 1992; Prach *et al.* 1997) during successional sequences. It was found that the majority of these traits exhibited patterns of distribution along the chronosequence, which could have been closely predicted from the earlier researchers findings. In addition, significant changes in trait representation of species in the vegetation along the chronosequence (e.g. seed longevity (declined); no. of competitors (increased); plant height (increased); leaf dry matter (increased)) could largely be attributed to the changing selection pressures usually associated with successional seres – i.e. the change in importance from colonising and regeneration potential at the outset of succession, towards competitive ability as a site ages. However, the results, which were of greatest interest, were those where these expected patterns of trait change along the chronosequence were not exhibited, most notably for seed weight (which did not increase) and mode of dispersal (which did not change from wind→animal). It is suggested that the key difference on urban derelict sites undergoing succession may be due to an increased occurrence of anthropochory. Seed weight and dispersal are typically, in natural

and semi-natural situations, correlated traits; animals are considerably more likely to be the **primary** dispersal mechanism for heavier seeds, whereas wind is more likely to be significant for small seeded species. However, anthropochory selects for a variety of seed sizes (Hodkinson & Thomson 1997), which can explain why the natural succession from large seeded to small seeded species is not as clear-cut in cities. The result is important as it provides quantitative evidence to support the assertion of previous authors (e.g. Gilbert 1989, Hill *et al.* 1994) that the significance of anthropochory in cities is very high.

The identification of different functional types, with different stages along the chronosequence, was also instrumental in demonstrating the increased specialisation in traits which occurs across the later successional subset of sites. On early successional sites, fewer functional groupings containing larger numbers of species were identified. Whereas across the older sites a greater number of functional groups were identified, some of which had very few members. This finding has implications for species diversity on these sites. Species diversity across sites, at the pioneer stage of succession, is always likely to be high, due to the stochastic nature of the factors determining whether a species successfully colonises. Important factors determining whether a pioneer takes hold on a site may be, for example, timing of denudation and proximity of a potential source population, as both of these will influence the ability to take advantage of the colonisation 'window'. The traits tested here, notably seed persistence and ruderality, are all important for their selection as pioneers, but other traits e.g. timing of their seed set, may be of greater significance when attempting to separate the relative likelihood of successful colonisation by a particular pioneer on a known site. Using the methods adopted here therefore it is important to acknowledge that the diversity of functional groupings observed at a particular stage in succession will be strongly determined by the nature of the traits analysed. Across older sites, a much greater degree of inevitability is incorporated into vegetation species composition as the community convergence shown in Chapter 2 demonstrated. Furthermore where variation does occur between or within sites it is not so much dependent on the stochastic factors influencing recruitment but upon environmental heterogeneity; differences in substrate are probably particularly important here. It is these environmental differences which lead to the increased functional diversity observed e.g. older topsoil sites are typified by competitive phanerograms whereas older sand and ballast sites have many low growing stress tolerators. The study

clearly demonstrates therefore, that diversity on and across young derelict sites is largely inherent, whereas diversity on and across older sites is dependent, upon environmental variability (mostly representation of different substrate types).

Objective 4: Examine the relative importance of environmental and spatial factors to determining the species assemblages found on sites, their species richness and their functional composition. A particular aim of the project was to determine the significance of different measures of habitat patch isolation and also to identify whether linear features enhanced connectivity in an otherwise fragmented landscape.

Environmental factors

Differences between the species assemblages recorded at sites could be most significantly attributed to variation in the Ellenberg environmental factors (light, moisture, pH and fertility) tested between sites. The representation of plant functional groups was also demonstrated to be significantly correlated with the Ellenberg environmental variables in Chapter 3 as were individual seed traits in Chapter 6. Substrate proved to also be an important variable in determining species composition, speed of succession and both site and quadrat species richness. In many instances environmental factors and substrate combined with the additional variables of age since dereliction and site size best predicted the majority of the explained variance in the data sets whereas spatial factors were often of limited significance.

Spatial features

1) Urban-Rural Gradient

No relationship was found between the extent of surrounding urban cover and the native species composition of the derelict flora, and nor was there was any significant transition observed in the distribution of native species along the gradient from the rural edge to the urban centres, (closely correlated with urban land cover) on these habitats. Factors that are typically associated with this gradient are increased habitat isolation, anthropogenic disturbance, pollution and temperature. However, further consideration of many of these

factors can lead to the conclusion that it is not at all surprising no native species gradient was demonstrated.

The results of this study have already demonstrated that habitat isolation may have some effect on the dispersal of some species, between sites in the urban zone. However this effect was not exhibited as an urban-rural gradient effect as well, because the density of derelict land in the West Midlands conurbation does not conform to any urban-rural gradient. If it did show a gradient it might have been expected to buck the trend of increasing habitat isolation, which is applicable to semi-natural woodlands or wetlands habitats, and in fact show less isolation towards the centre. In fact the largest hot spot for derelict land is in the north-west Black Country region with a smaller area of high occurrence towards the south-east (Figure 6.1). The centre of the conurbation is in fact marked by the large Sandwell Valley nature reserve made up of a woodland, scrub, grassland and aquatic habitat mosaic.

Anthropogenic disturbance is typical of many derelict sites and there is little evidence to suggest that the intensity or frequency of these events is likely to be higher towards the urban centre. In fact sites which are likely to be prone to most disturbance will typically be those either adjacent or close to residential areas. Density of housing, as with derelict land, varies across the West Midlands conurbation, and also does not conform to a straightforward gradient from edge to centre.

The existence of a clear urban-rural gradient for pollutant levels is also dubious. The remaining areas of industry in the conurbation are also clustered, and focused away from the geographical centres. Pollution from traffic is possibly now the major issue in the West Midlands, and nitrous oxide, carbon monoxide and sulphur dioxide levels can be expected to certainly be present at high levels in the urban centres. However the conurbation's motorways the M6, M5 and M42 are major intersections for traffic bypassing as well as entering the region. High levels of these gases can also therefore be expected along these transport links which pass through the outskirts and the centre of the conurbation.

Urban heat island effects have now been studied in major cities throughout the world (e.g. Brazdil & Budikova 1999, Kim & Baik 2002, Streutker 2002) including the West Midlands (Chapman, Thornes & Bradley 2002). This gradient may be a partial explanatory factor for the larger number of alien taxa present towards the urban centre, which was the major

significant species composition effect of urban cover, and the urban-rural gradient, demonstrated in this study. Similar findings have been made in central Europe (Pysek 1998), but this thesis presents the first quantitative assertion of this for the UK. Temperature may be an environmental control on the spread of many taxa introduced into the UK, as many will not be able to set seed in the summer, or are sensitive to frosts in winter. However the urban distribution was evident only in neophytes, and not represented at all by archaeophyte species, many of which originate from the Mediterranean region and might also be expected to prosper under hotter and dryer conditions typical of cities. . This suggests that the point of origin may be the most significant factor explaining this trend. Urban zones have a high density of gardens, from where recent neophyte escapes have often originated. City centres are also typically focal points of transport links, along which alien taxa often spread (e.g. Wilcox 1989; Lonsdale & Lane 1994). Consequently concentrations of these species might be expected towards urban centres. In contrast, archaeophytes were introduced into rural areas originally and have spread successfully over time into towns but are not significantly more prevalent there.

2) Habitat isolation

The study did demonstrate, very clearly, that sites in high-density areas of derelict land were likely to share more common species. However, this factor had no significant impact on the species richness, or the representation of traits in general although heavy seeded species of the Fabaceae were also found in significantly higher numbers at these sites. Interestingly no other possible measures of isolation had a significant impact on the similarity of species composition of sites, including site size, geographical distance between paired sites and density of urban cover. A possible explanation is that these high-density zones are currently subject to considerable ongoing demolition and redevelopment. There is consequently a fairly regular flow of construction vehicles such as bulldozers to and from different sites. It may be that, in these zones of high activity, this method of anthropogenic dispersal gives species from nearby derelict sites a potential head start, in colonising any new site. This would simply reduce the extent to which recruitment of a site is a lottery (*sensu* Sale 1977), but would not be expected to have an overall impact on species richness, which does not appear to be restricted by recruitment limitation on these sites in the long term. Indeed, there

is little consistent evidence from this study to suggest that habitat isolation has any significant impact on species in derelict habitats. This may be because these habitats are typically ephemeral, and select for highly dispersive species, and as such are likely to exhibit less significant changes under increased isolation than other urban habitats such as remnant woodland or wetland sites. It should also be acknowledged that determining habitat isolation for derelict habitats is very difficult in a complex urban mosaic, where analogous disturbed habitats such as road verges and untended gardens are ubiquitous. It may simply be that no site is truly isolated from a considerable local source species pool. There is evidently considerably more room for research into this aspect of urban ecology that should focus now on remnant semi-natural or natural habitats where isolation may be, firstly, more easily quantified and secondly, may be expected to have a more significant effect

3) Linear features

The evidence against the benefits of wildlife corridors as a means of increasing connectivity for isolated plant species and preserving or enhancing local diversity in cities was reasonably conclusive. A comprehensive review of the literature on this subject was carried out which clarified that, at present, there is little supportive field research data that corridors provide any mitigation for isolated plant populations. Furthermore it noted that these urban linear features do not fit the definition of true corridors i.e. they do not represent homogeneous habitat strips. Indeed worse still the review suggested that rather than enhancing native local species diversity, the weight of scientific evidence suggests that these features may be more of a threat by accelerating the spread of alien invasive taxa.

The research work in this thesis provides substantial additional quantitative evidence to back up the conclusions of the literature review. The proximity of urban linear features was demonstrated to have no affect on i) the overall similarity in floristic composition of sites, ii) species richness at site or quadrat level, iii) presence of rare or local species, or iv) proportion of poor dispersers (e.g. heavy seeded / zoochorous species). The study did demonstrate however that alien taxa formed the great majority of the few species that were associated more commonly with sites near corridors.

The implications of these findings are clear. Planners should stop attributing heightened biodiversity benefits to these features, something, which has become commonplace in the last decade, and nor should they prioritise them from a nature conservation perspective but rather evaluate them on the same basis as other non-linear fragments. It may well remain desirable to maintain linked green areas in cities, for recreational purposes or indeed from environmental viewpoints e.g. sustainable transport routes, and, in future, proposals should be justified on these grounds. Planners may also wish to pay greater consideration to the potential future costs of invasive plant control along these features when reaching their decisions.

9.3 A CONSERVATION STRATEGY FOR DERELICT LAND IN THE WEST MIDLANDS

The species diversity recorded across the derelict land survey is a significant component of the regional flora and as such should be considered in any strategic nature conservation proposals put forward by local authorities. However species diversity is inherently a product of the array of past land uses which occurred in the region and just as importantly even if, less predictably, it is also a product of activities by city dwellers such as tipping of waste and fire starting. Management strategies and plans fail to account for, and incorporate, such unpredictable events and have a tendency when applied consistently and repeatedly to achieve uniformity rather than diversity. It should be with reluctance therefore that ecologists offer planners a clearly defined solution to 'preserving' diversity at regional scales. Nevertheless, the current intensity of housing pressure suggests that the area of derelict land is going to undergo a yearly net loss to development for some considerable time. We therefore need to consider on what ecological basis site preservation decisions are made and afterwards how we should maintain these sites.

The first step in this process should be botanical survey of derelict sites. Botanical surveys of Nottingham (Shepherd 1992) and this study of the West Midlands highlight the significance of derelict land flora as an important component of urban diversity. Studies of other taxa on similar habitats have also found similar findings (e.g. Gilbert 1989 many taxa; Eversham *et al.* 1996 carabid beetles; Gibson 1998; several invertebrate groups). In addition the increasing knowledge about the distribution of our biodiversity which is resulting from the

number of new local Atlases being published (e.g. Birmingham and Black Country flora project in progress) should also be made available to environmental and planning division preferably in formats that enable site based searches.

The most important derelict sites, either in terms of their species richness or, equally importantly from a regional perspective, their distinctiveness, should be targeted for designation and appropriate management applied to ensure pioneer communities are retained. Good examples of protected former derelict land already exist in the West Midlands. Doulton Clay Pits near Dudley in the Black Country is designated as a Site of Special Scientific Interest (SSSI) and is managed within the Saltwells Local Nature Reserve. More recently, an old brickworks site later used as landfill near Sparkbrook only a couple of miles from Birmingham city centre has been designated as a local Site of Importance for Nature Conservation (SINC) and is managed as sections of woodland, wetland and open wet grassland. Extraction vents are used to release any dangerous gas accumulations.

At both these sites public usage is high and the areas are popular with locals. However, public support for maintaining derelict land as green space is often not present with less aesthetically sites. These are typically either i) those which are recently denuded and where vegetation development is slow because unfavourable substrate; or ii) those where conversely the vegetation has developed to scrub and rank grassland without any management. In the first case, the sites are undesirable as they are viewed typically as an eyesore, and as potential sites for fly-tipping or for travellers to set down their caravans. In the second case the tipping problem is also an issue but perhaps more significantly there is an element of fear because the site are viewed as potential locations for muggings and other crimes.

At larger sites there is certainly a lot of potential to implement measures, which will relieve some of the public pressure against derelict land. The use of gypsy banks and other measures to counter tipping and vehicular access at the edges of bare sites can be used in combination with temporary plantings as visual shields to counter the initial aesthetic problems of recently derelict sites. Cycleways and footpaths across these sites can also enable urban dwellers to appreciate the open space away from the roads. If these are accompanied with sufficient street lighting and sensibly routed away from more developed vegetation then the general public at large may be confident enough to use them.

In areas of the highest derelict density where land prices are by definition typically low, local authorities could aim for a strategic link up of derelict spaces. A network of open space could be created which with clever landscaping could retain areas of bare and regular disturbed ground while incorporating a mosaic of other more visually attractive habitats that dominate the eyeline. However, rather than considering networks as something which benefit people and wildlife, planners need to adopt a different, more ecologically justifiable, perspective. Networks should be envisaged and the routes through them landscaped with the considerations of city dwellers in mind. Rather than placing the emphasis on the network as providing conservation credit in its own right planners have to also manage the habitats within the network for biological diversity. Existing linear features may be interlinked within these networks but again care should be made that it is clear this is being done for the benefit of human use and not as mitigation for loss of valuable isolated habitats.

It is probably not possible for all of the problematic issues that confront planners when making decisions about derelict land to be resolved in all circumstances. Individual sites pose different questions and considerable thought and inspiration may be needed if ecologically sensitive, yet publicly acceptable answers are to be achieved. Yet the intensive phase of housing development, which we are beginning to experience, will probably rival that of the 1950s, which saw the clearance of so many old bombsite plant communities, and derelict land is surely on the decline. In light of the continued degradation which has taken place to our semi-natural and natural communities in the these last five decades it is surely worth the effort to preserve as much of these interesting, diverse and educational derelict habitats as we can.

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APPENDIX I

Details of the location, previous use, substrate type and history of the fifty derelict sites studied

SITE ID	SITE NAME	NGR	PREVIOUS USE	SUBSTRATE	OTHER NOTES
1	Kenyan Close	SO901851	Open land	Soil	Playing fields were sold to housing development in mid-1980s. Houses built 1986-7, and this space denuded then left "green". Mown regularly; Occasional dumping.
2	Frankley	SO999782	Unknown	Clay & hardcore	Area empty for >10years, then bulldozed in 1997 and left.
3	Erdington Large	SP131916	Houses	Brick rubble	Housing development cleared in 1996, site graded and left.
4	Erdington Small	SP120911	Houses	Brick rubble	Housing development cleared in 1996, site graded and left.
5	Glasscutters	SO894863	Factory	Broken Tarmac	Factory at rear of site derelict for several years, then burned and later demolished 1997. Front part bare and used as car lot. Disturbed by water works for pub (96/97) and garage (98/99)
6	Platts Road	SO895858	Houses	Brick rubble	Site disturbed when road was widened in 1996, then left.
7	Grange Road	SP138813	Verge	Broken Tarmac	Originally bushes/frontage of large house. Disturbed (bulldozed) when new flats built 1995/96. No rubbish. Partially turfed 2000. Mowing.
8	Solihull Arable	SP168818	Arable	Agricultural	Field was set-aside in 1995 (CityView)
9	Ackers	SP102844	Refuse Tip	Refuse & rubble	Derelict site (including tip) landscaped into ski slope 1993
10	Sports Centre	SP046838	Car Park	Broken Tarmac	Major disturbance in 1994 when new building constructed
11	Rubery shops	SO990784	Heap	Clay & hardcore	Possibly contaminated. Bulldozed in part in 1998 during investigative digging work: trees destroyed. Possibly housing >20 years ago
12	Five Ways	SP059857	Verge	Cinder & Ballast	Regularly disturbed / mown bank created with railway from ballast
13	Kings Heath pub	SP078796	Car Park	Broken Tarmac	Disused car park abandoned approximately 5 years ago
14	Little Aston	SK092004	Arable	Agricultural	Set-aside land from last cultivated 1996
15	Florence Road	SP030875	Houses	Brick rubble	Houses lay derelict from late eighties, later demolished in 1991/92.
16	Blake Street	SK104007	Open land	Soil	Agricultural land abandoned 1984/5 when rest of area was developed. Mown in late 1980s. Saplings cut May 1998 & bulldozed early 1999. .
17	Upper Reservoir Rd	SP099917	Houses	Brick rubble	Prefab housing became derelict in 1970-80 then demolished 1981-1983 Fire and tipping of garden waste.

Appendix 1 Continued

SITE ID	SITE NAME	NGR	PREVIOUS USE	SUBSTRATE	OTHER NOTES
18	Lower Reservoir Rd	SP104916	Houses	Brick rubble	Prefab housing became derelict in 1970-80 then demolished 1983-1985 Fire and tipping of garden waste.
19	Percy Road	SP097839	Houses	Brick rubble/ Tarmac	Houses derelict 1986-90, then garden centre 1991-94, then derelict 1995-1998. Bulldozed, fire, fly-tipping. New houses built 1998
20	Cradley Heath	SO949842	Garden	Garden soil	House demolished 1981-1985. Tipping of garden waste.
21	Minworth Verge	SP189924	Road edge	Soil	Created c. 1980 when new road was built.
22	Minworth Sewage	SP180920	Sewage	Sewage!	Regularly disturbed verge in sewage plant.
23	Small Bentley Mill	SO990979	Open land	Clay & hardcore	Raised bank created during building of M6, later disturbed during construction of drainage works.
24	Large Bentley Mill	SO988978	Sports Ground	Refuse / Soil	Sports ground closed 1970s. Tyre dumping in 1980s, fly-tipping, rabbits, motorbikes, freq. fires (tyres), Dog-walkers.
25	Hall Green Road	SP009944	Refuse Tip	Refuse/Broken Tarmac	Sand quarry until 1970s then refuse tip until 1990. Then capped with rubble and earth. "Most toxic site in W. Midlands". Some parts probably older than 1990. Frequent fires, tipping, oil.
26	Sandy Lane	SP013951	Farm	Soil	Abandoned farm land. Disturbances from grazing, fly-tipping & fires.
27	Vincent Drive	SP042830	Refuse Tip	Refuse	Industrial buildings & refuse tip until mid 1980s, then abandoned.
28	Crest View	SP087795	House	Broken Tarmac	Derelict house and garage bulldozed in late 1980s. Fly-tipping and fires
29	Saltwells Bus Stop	SO936878	Heath 1968	Soil	Dumping of waste, fire and disturbance by motorcross-bikes
30	Saltwells Verge	SO932874	Open land	Road gravel	Verge created at time of road widening
31	Soho Loop	SP050872	Railway sidings	Brick rubble /Ballast	One half was a disused carriage shed demolished c. 1988. Other was railway sidings, abandoned c. 1994.
32	Cole Bank Road	SP099817	Houses	Garden soil/ broken tarmac	Derelict houses and gardens, demolished in mid 1980s.
33	Samson Quarry	SO977891	Quarry	Sandy	Quarry closed 1960, filled in during 1970s. Garden waste, motorbikes.
34	Turners Hill	SO969880	Infilled workings	Clay & hardcore	Tarmac quarry/stone crushing workshops closed 1986/90. Some areas used for landfill. Disturbed by trucks, tipping, motorbikes, fire."
35	Burberry	SP099837	Brickwork	Clay & hardcore	Abandoned brickworks used as landfill then capped with clay

Appendix 1 Continued

SITE ID	SITE NAME	NGR	PREVIOUS USE	SUBSTRATE	OTHER NOTES
36	Roundhay	SP141875	Garages	Topsoil	Garages became derelict in 1981-85 and were cleared 1991-92 - top soil laid and shrubs planted. Fly-tipping, garden waste, fires.
37	Mid-Cole	SP132880	Open land	Topsoil	Land used for tipping. River dredgings dumped in 1970s. Fires & tipping.
38	Bearwood	SP035867	Railway	Sandy	Site by former bridge on disused railway. Sand dumped on top c. 1994 probably when nearby houses were built.
39	Wilson Road	SP026872	Houses	Brick rubble	Houses demolished on lower part in 1990, and on upper part in 1992. Fly-tipping, tipping of garden waste.
40	Ashted Circus	SP080874	Buildings	Broken tarmac/refuse	Factory buildings demolished 1993. Disturbance from lorries. Flytipping.
41	Woodlands Drive	SP007895	Houses/Works	broken tarmac / Garden soil	Factory buildings demolished 1996. Flytipping, garden waste, fires.
42	Foxyards Road	SO945925	Empty land	Refuse/Garden soil	Previously playing field, probably bulldozed when B'ham New Road was widened (pipes etc stored there). Fly tipping and garden waste. Rats.
43	Tunnel Street	SO942940	Houses	Broken Tarmac	Garage closed in 1980s after fire, then demolished probably around 1990 and bulldozed by Council later 1990s. Tipping & disturbance from trucks
44	Old Park Road	SO981964	School	Broken Tarmac	School burnt down in late 1980s and was later cleared (early 1990s). Fly-tipping, garden waste, fire.
45	Mounts Road	SO988945	Houses	Brick /Garden soil/ Tarmac	Flats derelict due to subsidence (1995/6) and bulldozed in May 1998. Garden waste, fly-tipping
46	Heath Street	SP044877	Houses	Brick rubble	Houses demolished 1970, toilets demolished 1995. Caravans on in about 1995, causing soil disturbance. Fly-tipping
47	Walsall	SP013991	Yard	Broken Tarmac	Factory buildings & yard demolished c. 1990. Garden waste, fly-tipping.
48	Institute Road	SP074817	Public baths	Brick rubble/Refuse	Swimming baths closed mid-1980s. Demolished and bulldozed 1986/88. Bank at edge of site was garden of swimming baths. Fly-tipping
49	Tyseley Wharf	SP113843	Buildings	Clay & hardcore	Bank created in 1995 during construction of nearby factory buildings.
50	Brownhills	SK033055	Railway	Cinder & Ballast	Railway closed in 1960s. Some tipping.

APPENDIX II

Phytosociological table of the most commonly recorded plant communities found on synanthropic urban locations in Britain. Adapted to incorporate the urban classifications of Shepherd with relevant NVC communities. Sources: Shepherd 1994, Rodwell ed. 1992, 2000.

CLASS	ORDER	ALLIANCE	COMMUNITY	DESCRIPTION OF URBAN LOCATIONS
GALIO-URTICETEA Passarge ex Kopecky 1969	LAMIO ALBI-CHENOPODIETALIA BONIHENRICI Kopecky	**Galio-Alliarion** (Oberdorfer 1957) Lohmeyer et Oberdorfer in Oberdorfer *et al.* 1967	OV24 *Urtica dioica-Galium aparine* community	Well-aerated nutrient rich soils on dumps of soil on wasteland or in unkept gardens
			OV25 *Urtica dioica-Cirsium arvense* community	Nutrient-rich loamy soils on verges and waste ground
ARTEMISIETEA VULGARIS Lohmeyer et al. Ex Rochow 1951	ONOPORDIETALIA ACANTHII Br.-Bl. Tx. Ex Klika & Hadac 1944	**Arction lappae** Tuxen 1937 em. Gutte 1972	SH6 *Lamio-Conietum maculati*	Rubbish tips & earth mounds & on disturbed ground in moist situations e.g. adjacent to canals or ditches
			SH7 *Tanaceto-Artemisietum*	Edges of derelict plots, car parks, & roadsides on dry poor to moderately fertile soil.
			SH9 *Artemisia vulgaris-Urtica dioica* community	Earth and rubble mounds on derelict building plots & on road verges. Sandy loams mixed with brick, concrete and rubbish
		Rumicion obtusifolii Gutte 1971	SH10 *Rumicietum obtusifolii*	Mounds of earth on construction sites or on sides of the road over sandy loams
MELILOTO-ARTEMISIETEA ABSINTHII Elias 1979	MELILOTO-ARTEMISIETEA ABSINTHII Elias 1980	**Dauco-Melilotion** Gutte 1972	SH11 *Melilotetum albae-officinalis*	On derelict land, clay pits & railway sidings. Sandy loams with brick rubble or concrete

Appendix II Continued

CLASS	ORDER	ALLIANCE	COMMUNITY	DESCRIP[N.] URBAN LOCATIONS
EPILOBIETEA ANGUSTIFOLII Tuxen et Preising ex van Rochow 1951	ATROPETALIA Vlieger 1937	**Sambuco-Salicion capreae** Tuxen & Neum.1950	SH12 Epilibio-Salicetum (Oberdorfer 1957)	Derelict sites over concrete & brick rubble often with thin dry gravelly soil
		Carici piluliferae-Epilobion angustifolii Tuxen 1950	OV27 *Epilobium angustifolium* community	Railway embankments, wasteland, old fire patches, over fertile soil.
STELLARIETEA MEDII Tuxen, Lohmeyer et Preising ex Rochow 1951	SISYMBRIETALIA J Tuxen in Lohmeyer *et al.* 1962	**Sisymbrium officinalis** Tuxen, Lohmeyer and Preising 1950 Not determined	SH1 *Hordeetum murinii*	Road verges, base of walls, edges of car parks and paved areas on dry soils
			SH2 *Sisymbrietum loeselii*	Brick & earth mounds, edges of car parks
			SH3 *Sisymbrium officinale-Tripleurospermum* community	Mounds of earth, brick and concrete often on construction sites over fertile sandy loams
			SH5 *Senecio squalidus-Conyza canadensis* community	Bare patches, base of walls, over brick rubble, gravel, concrete & cinder track. Well drained, dry soil with low organic matter
	NOT DETERMINED	**Not determined**	SH4 *Chenopodium-Atriplex prostata* community	Mounds of earth with brick rubble & given topsoil application. Moist fertile soils.
	POLYGONO-CHENOPODIETALIA R.Tx. et Lohmeyer 1950 em. J.Tx. 1961	**Polygono-Chenopodion polyspermii** W Koch 1926 em Sissingh 1946	OV 9 *Matricaria perforata-Stellaria media* community	Road sides
			OV 10 *Poa annua-Senecio vulgaris* community	Trampled ground, on dumped earth, gardens, recreation fields
			OV12 *Poa annua-Myosotis arvensis* community	Trampled recreational areas

Appendix II Continued

CLASS	ORDER	ALLIANCE	COMMUNITY	DESCRIP[N.] URBAN LOCATIONS
STELLARIETEA MEDII Tuxen, Lohmeyer et Preising ex Rochow 1951 **(cont)**	CENTAUREETALIA CYANI R.Tx.,Lohmeyer et Preising in R.Tx. 1950	**Fumario-Euphorbion** Th.Muller ex Gors 1966	SH5 *Fumaria officinalis-Veronica persica* community	Predominantly in landscape areas growing over freshly laid topsoil
			OV13 *Stellaria media-Capsella bursa-pastoris* community	Disturbed ground enriched through dumping of topsoil
MOLINIO-ARRHENATHERETEA Tuxen 1937	MOLINIETALIA CAERULEAE Koch 1926	**Calthion palustris** Tuxen 1937 em. Balatova-Tulakova 1978	MG9 *Holcus lanatus-Deschampsia cespitosa* grassland	Waste ground most frequently where soil conditions are anaerobic due to waterlogging
	ARRHENATHERETALIA Tuxen 1926	**Arrhenatherion elatioris** Koch 1926	MG1 *Arrhenatherum elatius* grassland / SH16 *Arrhenatheretum elatioris* Braun Blaunquet 1919 **	Widespread. Road verges, embankments, churchyards, waste ground, dumps, disused quarries on a variety of substrata
		Cynosurion cristati Tuxen 1947	MG5 *Cynosurus cristatus-Centaurea nigra* grassland	Road verges, railway embankments a&old quarries
			MG6 *Lolium perenne-Cynosurus cristatus* grassland	Recreation swards, lawns & road verges on moist free draining soils
POLYGONO ARENASTRI-POETEA ANNUAE Rivas-Martinez 1975 corr. Rivas-Martinez et al. 1991	POLYGONO ARENASTRI-POETALIA ANNUAE R.Tx. in Gehu et al. 1972 corr. Rivas-Martinez et al. 1991	**Lolio-Plantaginion** Sissingh 1960	MG7 *Lolium perenne* leys and related grasslands	Recreational areas & damp road verges & habitats prone to light trampling
			OV21 *Poa annua-Plantago major* community	Heavily trampled areas such as tracks & paving in recreation areas, wasteland a& road sides
			OV22 *Poa annua-Plantago major community*	Disturbed, lightly trampled paths, beside walls, pavements, verges, gardens & wasteland

Appendix II Continued

CLASS	ORDER	ALLIANCE	COMMUNITY	DESCRIP[N.] URBAN LOCATIONS
POLYGONO ARENASTRI-POETEA ANNUAE Rivas-Martinez 1975 corr. Rivas-Martinez et al. 1991 **(cont)**	POLYGONO ARENASTRI-POETALIA ANNUAE R.Tx. in Gehu et al. 1972 corr. Rivas-Martinez et al. 1991 **(cont)**	**Lolio-Plantaginion** Sissingh 1960 **(cont)**	OV23 *Poa annua-Taraxacum officinale* community	Sown recreation areas, playing fields, some disturbance & low frequency mowing
		Polgonion avicularis Br.-Bl. Ex Aichinger 1933	OV18 *Polygonum aviculare-Chamomilla suaveolens* community	Paths over loamy and sandy soils on wasteground & lightly trampled recreational areas
			OV19 *Poa annua-Matricaria perforata* community	Road verges
			OV20 *Poa annua-Sagina procumbens* community	Cracks in well kept pavements and walls, often the only vegetation type where present
			SH13 *Polygono-Matricaria matricarioidis* Sissingh 1969 em Tuxen 1972	Road verges, gardens, landscaped areas prone to disturbance. Fertile loamy sands.
		Elymo-Rumicion crispi Nordhagen 1940	MG11 *Festuca rubra-Agrostis stolonifera-Potentilla anserina* grassland	Moist but free draining soils alongside ditches and roadsides
			OV28 *Agrostis stolonifera-Ranunculus repens* community	Waterlogged clay waste ground and muddy tracks
		Thero-Airion Tuxen 1951	SH14 *Hieracium pilosella* community	Railway lines on banks and cinder and clinker.
			SH15 *Vulpia myuros* community	Disused railway lines on loose dry soils with cinder and clinker
RHAMNO-PRUNETEA Rivas Goday et Borja Carbonell 1961	PRUNETALIA SPINOSAE Tuxen 1952	**Rubion subatlanticum** R.Tx. 1952	W24 *Rubus fruticosus-Holcus lanatus* understorey	Abandoned and neglected land particularly derelict sites and disused gardens on a variety of substrata

APPENDIX III

Matches between NVC and Shepherd communities and those recorded during the study.

The following tables compare species present in quadrats to the community scores they were shown to exhibit a best match to in Chapter 2. Quadrats which provided a match of 50 or above in the TABLEFIT output to the same community were aggregated to provide total incidence number for each species. The frequency of each species within these combined quadrats was then derived by dividing by the total number of matches (in brackets). These are compared to the expected cover scores for the respective NVC or Shepherd community.

MG1 Arrhenatherum elatius grassland / SH16 Arrhenatheretum elatioris

Species	Cover Score	Quad (105)		Species	Cover Score	Quad (105)
Arrhenatherum elatius	V	100%		*Agrostis capillaris*	I	16%
Dactylis glomerata	IV	44%		*Rumex obtusifolius*	I	13%
Holcus lanatus	III	56%		*Cerastium fontanum*	I	11%
Plantago lanceolata	III	50%		*Geranium dissectum*	I	10%
Cirsium arvense	III	39%		*Potentilla reptans*	I	8%
Achillea millefolium	III	21%		*Cirsium vulgare*	I	6%
Urtica dioica	III	16%		*Ranunculus acris*	I	6%
Centaurea nigra	III	11%		*Ranunculus repens*	I	6%
Heracleum sphondylium	III	8%		*Vicia cracca*	I	6%
Festuca rubra	II	47%		*Leucanthemum vulgare*	I	5%
Rubus fruticosus agg.	II	36%		*Anisantha sterilis*	I	4%
Taraxacum	II	27%		*Epilobium hirsutum*	I	4%
Senecio jacobaea	II	25%		*Hypochaeris radicata*	I	3%
Poa pratensis	II	20%		*Phleum pratense*	I	3%
Lolium perenne	II	17%		*Rumex crispus*	I	2%
Lotus corniculatus	II	15%		*Vicia sepium*	I	2%
Trifolium pratense	II	15%		*Conopodium majus*	I	2%
Vicia sativa	II	13%		*Convolvulus arvensis*	I	2%
Galium aparine	II	11%		*Pimpinella saxifraga*	I	2%
Poa trivialis	II	10%		*Pteridium aquilinum*	I	2%
Trifolium repens	II	8%		*Rhinanthus minor*	I	2%
Elytrigia repens	II	7%		*Festuca pratensis*	I	1%
Lathyrus pratensis	II	6%		*Prunella vulgaris*	I	1%
Rumex acetosa	II	3%		*Sonchus asper*	I	1%
Anthriscus sylvestris	II	3%		*Aegopodium podagraria*	I	1%
Galium verum	II	0%		*Alopecurus pratensis*	I	1%
Pastinaca sativa	II	0%		*Cynosurus cristatus*	I	1%
Agrostis stolonifera	I	56%		*Hypericum perforatum*	I	1%
Artemisia vulgaris	I	22%		*Origanum vulgare*	I	1%
Chamerion angustifolium	I	17%		*Stellaria graminea*	I	1%

MG1 / SH16 Continued...

Species	Cover Score	Quad (105)		Species	Cover Score	Quad (105)
Agrimonia eupatoria	I	0%		*Linum catharticum*	I	0%
Anthoxanthum odoratum	I	0%		*Lotus uliginosus*	I	0%
Avenula pubescens	I	0%		*Luzula campestris*	I	0%
Brachypodium sylvaticum	I	0%		*Papaver dubium*	I	0%
Briza media	I	0%		*Papaver rhoeas*	I	0%
Bromopsis erecta	I	0%		*Pimpinella major*	I	0%
Bromus hordeaceus	I	0%		*Primula veris*	I	0%
Campanula rotundifolia	I	0%		*Ranunculus bulbosus*	I	0%
Capsella bursa-pastoris	I	0%		*Ranunculus ficaria*	I	0%
Carduus nutans	I	0%		*Sanguisorba minor*	I	0%
Centaurea scabiosa	I	0%		*Scabiosa columbaria*	I	0%
Clinopodium vulgare	I	0%		*Silene nutans*	I	0%
Daucus carota	I	0%		*Symphytum officinale*	I	0%
Festuca ovina	I	0%		*Teucrium scorodonia*	I	0%
Filipendula ulmaria	I	0%		*Thymus praecox*	I	0%
Galium mollugo	I	0%		*Trisetum flavescens*	I	0%
Glechoma hederacea	I	0%		*Veronica chamaedrys*	I	0%
Hedera helix	I	0%		*Crepis capillaris*	-	19%
Helianthemum nummularium	I	0%		*Vicia hirsuta*	-	18%
Holcus mollis	I	0%		*Medicago lupulina*	-	13%
Knautia arvensis	I	0%		*Crataegus monogyna*	-	11%
Lamium album	I	0%		*Linaria vulgaris*	-	10%
Leontodon hispidus	I	0%		*Tussilago farfara*	-	10%

MG5 Cynosurus cristatus-Centaurea nigra grassland

Species	Cover Score	Quad (17)	Species	Cover Score	Quad (17)
Festuca rubra	V	78%	*Alchemilla xanthochlora*	I	0%
Lotus corniculatus	V	78%	*Alopechorus pratensis*	I	0%
Plantago lanceolata	V	78%	*Avenula pubescens*	I	0%
Cynosurus cristatus	V	0%	*Bromus hordeaceus*	I	0%
Holcus lanatus	IV	89%	*Cardamine pratensis*	I	0%
Trifolium pratense	IV	67%	*Carex caryophyllea*	I	0%
Agrostis capillaris	IV	44%	*Carex flacca*	I	0%
Anthoxanthum odoratum	IV	11%	*Carex panicea*	I	0%
Centaurea nigra	IV	11%	*Colchicum autumnale*	I	0%
Dactylis glomerata	IV	11%	*Conopodium majus*	I	0%
Trifolium repens	IV	11%	*Danthonia decumbens*	I	0%
Ranunculus acris	III	78%	*Festuca arundinacea*	I	0%
Hypochaeris radicata	III	56%	*Festuca ovina*	I	0%
Prunella vulgaris	III	44%	*Festuca pratensis*	I	0%
Achillea millefolium	III	11%	*Filipendula ulmaria*	I	0%
Leontodon autumnalis	III	11%	*Juncus articulatus*	I	0%
Lolium perenne	III	11%	*Juncus effusus*	I	0%
Rumex acetosa	III	11%	*Juncus inflexus*	I	0%
Taraxacum agg.	III	11%	*Knautia arvensis*	I	0%
Ranunculus bulbosus	III	0%	*Koeleria macrantha*	I	0%
Trisetum flavescens	III	0%	*Luzula campestris*	I	0%
Cerastium fontanum	II	44%	*Ophioglossum vulgatum*	I	0%
Leucanthemum vulgare	II	44%	*Phleum pratense*	I	0%
Primula veris	II	33%	*Pimpinella saxifraga*	I	0%
Cirsium arvense	II	22%	*Plantago media*	I	0%
Galium verum	II	22%	*Potentilla erecta*	I	0%
Lathyrus pratensis	II	11%	*Silaum silaus*	I	0%
Poa pratensis	II	11%	*Stachys betonica*	I	0%
Rhinanthus minor	II	11%	*Succisa pratensis*	I	0%
Arrhenatherum elatius	II	0%	*Vicia cracca*	I	0%
Bellis perennis	II	0%	*Cynosurus cristatus*	-	89%
Briza media	II	0%	*Daucus carota*	-	56%
Heracleum sphondylium	II	0%	*Medicago lupulina*	-	44%
Leontodon hispidus	II	0%	*Crataegus monogyna*	-	33%
Poa trivialis	II	0%	*Anthyllis vulneraria*	-	22%
Trifolium dubium	II	0%	*Pastinaca sativa*	-	22%
Veronica chamaedrys	II	0%	*Rubus fruticosus agg.*	-	22%
Agrostis stolonifera	I	56%	*Cirsium vulgare*	-	11%
Senecio jacobaea	I	44%	*Hieracium*	-	11%
Crepis capillaris	I	22%	*Leontodon saxatilis*	-	11%
Potentilla reptans	I	11%	*Ligustrum vulgare*	-	11%
Ranunculus repens	I	11%	*Salix caprea*	-	11%
Sanguisorba minor	I	11%	*Sonchus asper*	-	11%
Agromonia eupatoria	I	0%	*Tragopogon pratensis*	-	11%
Alchemilla filicaulis	I	0%	*Tussilago farfara*	-	11%
Alchemilla glabra	I	0%			

MG6 Lolium perenne-Cynosurus cristatus grassland

Species	Cover Score	Quad (17)		Species	Cover Score	Quad (17)
Lolium perenne	V	100%		*Deschampsia cespitosa*	I	6%
Trifolium repens	V	100%		*Hypochaeris radicata*	I	6%
Cynosurus cristatus	V	56%		*Medicago lupulina*	I	6%
Holcus lanatus	IV	75%		*Plantago major*	I	6%
Agrostis capillaris	III	63%		*Poa annua*	I	6%
Festuca rubra	III	56%		*Potentilla reptans*	I	6%
Taraxacum agg.	III	44%		*Trisetum flavescens*	I	6%
Ranunculus acris	III	6%		*Alopecurus geniculatus*	I	0%
Cerastium fontanum	III	0%		*Cardamine pratensis*	I	0%
Dactylis glomerata	III	0%		*Centaurea nigra*	I	0%
Agrostis stolonifera	II	63%		*Festuca ovina*	I	0%
Plantago lanceolata	II	31%		*Festuca pratensis*	I	0%
Ranunculus repens	II	31%		*Iris pseudacorus*	I	0%
Cirsium arvense	II	25%		*Juncus effusus*	I	0%
Cirsium vulgare	II	25%		*Lathyrus pratensis*	I	0%
Trifolium pratense	II	19%		*Leontodon autumnalis*	I	0%
Poa trivialis	II	13%		*Luzula campestris*	I	0%
Rumex acetosa	II	13%		*Potentilla anserina*	I	0%
Achillea millefolium	II	6%		*Prunella vulgaris*	I	0%
Poa pratensis	II	6%		*Rhinanthus minor*	I	0%
Alopechurus pratensis	II	0%		*Trifolium dubium*	I	0%
Bellis perennis	II	0%		*Veronica chamaedrys*	I	0%
Phleum pratense	I	31%		*Crepis capillaris*	-	31%
Arrhenatherum elatius	I	19%		*Artemisia vulgaris*	-	19%
Rumex obtusifolius	I	19%		*Sonchus asper*	-	19%
Anthoxanthum odoratum	I	6%		*Chamerion angustifolium*	-	13%
Bromus hordeaceus	I	6%		*Cytisus scoparius*	-	13%

MG7 Lolium perenne leys and related grasslands

Species	Cover Score	Quad (17)		Species	Cover Score	Quad (17)
Lolium perenne	V	100%		*Poa annua*	I	6%
Trifolium repens	V	59%		*Alopecurus pratensis*	I	0%
Dactylis glomerata	IV	59%		*Bellis perennis*	I	0%
Phleum pratense	IV	29%		*Capsella bursa-pastoris*	I	0%
Holcus lanatus	III	59%		*Cerastium glomeratum*	I	0%
Agrostis capillaris	III	41%		*Hypochaeris radicata*	I	0%
Cerastium fontanum	III	18%		*Juncus inflexus*	I	0%
Taraxacum agg.	II	59%		*Ranunculus bulbosus*	I	0%
Cirsium arvense	II	29%		*Ranunuculus acris*	I	0%
Poa trivialis	II	29%		*Rumex acetosa*	I	0%
Vicia sativa	II	29%		*Rumex conglomeratus*	I	0%
Ranunculus repens	II	18%		*Rumex crispus*	I	0%
Rumex obtusifolius	II	18%		*Stellaria media*	I	0%
Achillea millefolium	II	6%		*Vicia cracca*	I	0%
Bromus hordeaceus	II	6%		*Artemisia vulgaris*	-	35%
Trifolium dubium	II	0%		*Crepis capillaris*	-	18%
Plantago lanceolata	I	47%		*Rubus fruticosus agg.*	-	18%
Festuca rubra	I	35%		*Sonchus asper*	-	18%
Cirsium vulgare	I	29%		*Vicia hirsuta*	-	18%
Poa pratensis	I	29%		*Geranium dissectum*	-	12%
Trifolium pratense	I	29%		*Leontodon autumnalis*	-	12%
Agrostis stolonifera	I	24%		*Medicago lupulina*	-	12%
Plantago major	I	24%		*Quercus robur*	-	12%
Arrhenatherum elatius	I	12%		*Solidago canadensis*	-	12%
Elytrigia repens	I	6%				
Festuca pratensis	I	6%				

MG9 Holcus lanatus-Deschampsia cespitosa grassland

Species	Cover Score	Quad (17)		Species	Cover Score	Quad (17)
Deschampsia cespitosa	V	90%		*Carex hirta*	I	0%
Holcus lanatus	IV	70%		*Carex nigra*	I	0%
Agrostis stolonifera	III	100%		*Carex panicea*	I	0%
Ranunculus repens	III	50%		*Cirsium palustre*	I	0%
Dactylis glomerata	III	20%		*Conopodium majus*	I	0%
Festuca rubra	III	20%		*Cynosurus cristatus*	I	0%
Poa trivialis	III	10%		*Dactylorhiza fuchsii*	I	0%
Plantago lanceolata	II	60%		*Festuca arandinacea*	I	0%
Arrhenatherum elatius	II	50%		*Galium verum*	I	0%
Cirsium arvense	II	50%		*Glyceria fluitans*	I	0%
Lolium perenne	II	30%		*Heraclium sphondylium*	I	0%
Alopecurus pratensis	II	20%		*Holcus mollis*	I	0%
Agrostis capillaris	II	10%		*Hordium secalinum*	I	0%
Centaurea nigra	II	10%		*Hypericum perforatum*	I	0%
Cerastium fontanum	II	10%		*Juncus articulatus*	I	0%
Juncus effusus	II	10%		*Leontodon autumnalis*	I	0%
Lathyrus pratensis	II	10%		*Leucanthemum vulgare*	I	0%
Anthoxanthum odoratum	II	0%		*Lotus uliginosus*	I	0%
Festuca pratensis	II	0%		*Mentha aquatica*	I	0%
Filipendula ulmaria	II	0%		*Phalaris arundinacea*	I	0%
Juncus inflexus	II	0%		*Potentilla anserina*	I	0%
Ranunculus acris	II	0%		*Potentilla erecta*	I	0%
Rumex acetosa	II	0%		*Primula veris*	I	0%
Vicia cracca	I	40%		*Prunella vulgaris*	I	0%
Taraxacum agg.	I	30%		*Pulicaria dysenterica*	I	0%
Trifolium pratense	I	30%		*Ranunculus ficaria*	I	0%
Lotus corniculatus	I	20%		*Rumex conglomeratus*	I	0%
Rubus fruticosus agg.	I	20%		*Rumex crispus*	I	0%
Rumex obtusifolius	I	20%		*Rumex sanguineus*	I	0%
Cirsium vulgare	I	10%		*Senecio aquaticus*	I	0%
Phleum pratense	I	10%		*Senecio jacobaea*	I	0%
Poa pratensis	I	10%		*Silaum silaus*	I	0%
Trifolium repens	I	10%		*Succisa pratensis*	I	0%
Urtica dioica	I	10%		*Trisetum flavescens*	I	0%
Achillea millefolium	I	0%		*Potentilla reptans*	-	40%
Achillea ptarmicha	I	0%		*Medicago lupulina*	-	30%
Agrimonia eupatoria	I	0%		*Tussilago farfara*	-	30%
Angelica sylvestris	I	0%		*Calystegia silvatica*	-	20%
Briza media	I	0%		*Elytrigia repens*	-	20%
Cardamine pratensis	I	0%		*Vicia sativa*	-	20%

MG11a Festuca rubra-Agrostis stolonifera-Potentilla anserina grassland

Species	Cover Score	Quad (105)		Species	Cover Score	Quad (105)
Agrostis stolonifera	V	100%		*Polygonum aviculare*	I	2%
Lolium perenne	V	92%		*Rumex crispus*	I	2%
Holcus lanatus	III	55%		*Urtica dioica*	I	2%
Trifolium repens	III	53%		*Potentilla reptans*	I	1%
Festuca rubra	III	40%		*Alopecurus geniculatus*	I	0%
Cerastium fontanum	III	12%		*Bromus hordeaceus*	I	0%
Potentilla anserina	III	0%		*Carex distans*	I	0%
Taraxacum	II	54%		*Cochlearia officinalis*	I	0%
Plantago lanceolata	II	33%		*Festuca arundinacea*	I	0%
Rumex obtusifolius	II	29%		*Glaux maritima*	I	0%
Ranunculus repens	II	21%		*Juncus articulatus*	I	0%
Agrostis capillaris	II	20%		*Juncus bufonius*	I	0%
Cirsium arvense	II	17%		*Juncus gerardi*	I	0%
Dactylis glomerata	II	15%		*Odontites verna*	1	0%
Festuca pratensis	II	0%		*Oenanthe lachenalii*	1	0%
Stellaria media	II	0%		*Plantago maritima*	1	0%
Plantago major	I	25%		*Ranunculus acris*	1	0%
Leontodon autumnalis	I	17%		*Triglochin maritimum*	1	0%
Poa trivialis	I	15%		*Tripleurospermum maritimum*	1	0%
Cirsium vulgare	I	14%		*Crepis capillaris*	-	20%
Phleum pratense	I	11%		*Trifolium pratense*	-	19%
Poa pratensis	I	11%		*Epilobium montanum*	-	17%
Arrhenatherum elatius	I	10%		*Artemisia vulgaris*	-	16%
Elytrigia repens	I	9%		*Medicago lupulina*	-	16%
Lotus corniculatus	I	7%		*Sonchus asper*	-	14%
Hypochaeris radicata	I	6%		*Vicia sativa*	-	13%
Galium aparine	I	5%		*Epilobium hirsutum*	-	10%
Bellis perennis	I	3%		*Sonchus oleraceous*	-	10%
Poa annua	I	3%				
Rumex acetosa	I	3%				
Atriplex prostrata	I	2%				
Matricaria discoidea	I	2%				

OV23 Poa annua-Taraxacum officinale community

Species	Cover Score	Quad (39)		Species	Cover Score	Quad (39)
Lolium perenne	V	85%		*Buddleja davidii*	I	3%
Taraxacum agg.	IV	69%		*Centaurea nigra*	I	3%
Dactylis glomerata	IV	62%		*Daucus carota*	I	3%
Plantago lanceolata	IV	49%		*Festuca ovina*	I	3%
Poa annua	III	21%		*Galium aparine*	I	3%
Achillea millefolium	III	13%		*Holcus mollis*	I	3%
Agrostis stolonifera	II	64%		*Matricaria discoidea*	I	3%
Medicago lupulina	II	56%		*Sonchus oleraceous*	I	3%
Holcus lanatus	II	51%		*Tragopogon pratense*	I	3%
Trifolium repens	II	38%		*Tripleurospermum inodorum*	I	3%
Arrhenatherum elatius	II	36%		*Tussilago farfara*	I	3%
Trifolium pratense	II	28%		*Vulpia myuros*	I	3%
Plantago major	II	23%		*Capsella bursa-pastoris*	I	0%
Rumex obtusifolius	II	23%		*Cerastium glomeratum*	I	0%
Vicia sativa	II	15%		*Chamerion angustifolium*	I	0%
Hypochaeris radicata	II	13%		*Cichorium intybum*	I	0%
Urtica dioica	II	13%		*Cymbalaria muralis*	I	0%
Ranunculus repens	II	10%		*Heracleum sphondylium*	I	0%
Bromus hordeaceous	II	5%		*Lapsana communis*	I	0%
Crepis vesicaria	II	25%		*Malva sylvatica*	I	0%
Trifolium dubium	II	0%		*Medicago arabica*	I	0%
Agrostis capillaris	I	13%		*Phleum pratense*	I	0%
Crepis capillaris	I	7%		*Picris echioides*	I	0%
Cirsium vulgare	I	21%		*Plantago coronopus*	I	0%
Poa trivialis	I	21%		*Potentilla reptans*	I	0%
Cirsium arvense	I	15%		*Ranunculus bulbosus*	I	0%
Artemisia vulgaris	I	13%		*Rumex crispus*	I	0%
Sonchus asper	I	13%		*Sagina procumbens*	I	0%
Festuca rubra	I	10%		*Senecio vulgaris*	I	0%
Geranium dissectum	I	10%		*Silene vulgaris*	I	0%
Poa pratensis	I	10%		*Spergula arvensis*	I	0%
Senecio jacobaea	I	10%		*Stellaria media*	I	0%
Senecio squalidus	I	10%		*Veronica arvensis*	I	0%
Anisantha sterilis	I	8%		*Acer pseudoplatanus*	-	5%
Cerastium fontanum	I	8%		*Rubus fruticosus*	-	21%
Cynosurus cristatus	I	5%		*Epilobium montanum*	-	15%
Hordeum murinum	I	5%		*Vicia hirsuta*	-	13%
Leucanthemum vulgare	I	5%		*Betula pendula*	-	10%
Sisymbrium officinale	I	5%		*Hieracium agg.*	-	10%
Bellis perennis	I	3%				

OV25 Urtica dioica-Cirsium arvense community

Species	Cover Score	Quad (39)		Species	Cover Score	Quad (39)
Urtica dioica	IV	63%		*Capsella bursa-pastoris*	I	0%
Cirsium arvense	IV	60%		*Carduus acanthoides*	I	0%
Elytrigia repens	III	53%		*Centaurea scabiosa*	I	0%
Arrhenatherum elatius	III	53%		*Daucus carota*	I	0%
Lolium perenne	III	23%		*Echium vulgare*	I	0%
Dactylis glomerata	III	17%		*Epilobium hirsutum*	I	0%
Holcus lanatus	II	50%		*Geranium molle*	I	0%
Galium aparine	II	50%		*Glechoma hederacea*	I	0%
Agrostis stolonifera	II	47%		*Hordeum murinum*	I	0%
Calystegia sepium	II	30%		*Leucanthemum vulgare*	I	0%
Rumex obtusifolius	II	30%		*Myosotis arvensis*	I	0%
Artemisia vulgaris	II	23%		*Phragmites australis*	I	0%
Cirsium vulgare	II	13%		*Potentilla reptans*	I	0%
Sonchus asper	II	13%		*Pteridium aquilinum*	I	0%
Anisantha sterilis	II	40%		*Senecio vulgaris*	I	0%
Poa annua	II	7%		*Silene dioica*	I	0%
Sisymbrium officinale	II	3%		*Trifolium pratense*	I	0%
Heracleum sphondylium	II	3%		*Senecio jacobaea*	I	0%
Lamium purpureum	II	0%		*Poa pratensis*	-	17%
Malva sylvestris	II	0%		*Agrostis capillaris*	-	13%
Rubus fruticosus	I	30%		*Buddleja davidii*	-	13%
Anthriscus sylvestris	I	13%		*Chamerion angustifolium*	-	13%
Ranunculus repens	I	13%		*Poa trivialis*	-	10%
Achillea millefolium	I	10%		*Taraxacum agg.*	-	10%
Conium maculatum	I	7%				
Plantago lanceolata	I	7%				
Cerastium fontanum	I	3%				
Dipsacus fullonum	I	3%				
Geranium dissectum	I	3%				
Sonchus oleraceous	I	3%				
Trifolium repens	I	3%				
Vicia sativa	I	3%				
Aster tripolium	I	0%				
Atriplex prostata	I	0%				
Avena fatua	I	0%				

OV27b Epilobium angustifolium community

Species	Cover Score	Quad (17)		Species	Cover Score	Quad (17)
Chamerion angustifolium	V	88%		*Dryopteris dilatata*	I	0%
Urtica dioica	IV	53%		*Dryopteris filix-mas*	I	0%
Rubus fruiticosus	III	59%		*Elytrigia repens*	I	0%
Holcus lanatus	III	35%		*Fraxinus excelsior*	I	0%
Cirsium arvense	III	24%		*Glechoma hederacea*	I	0%
Arrhenatherum elatius	II	41%		*Hedera helix*	I	0%
Galium aparine	II	29%		*Hyacynthoides non-scripta*	I	0%
Poa trivialis	II	24%		*Juncus conglomeratus*	I	0%
Agrostis capillaris	II	12%		*Juncus effusus*	I	0%
Cirsium vulgare	II	12%		*Lathyrus pratensis*	I	0%
Heracleum sphondylium	II	6%		*Mercurialis perennis*	I	0%
Dactylis glomerata	II	0%		*Potentilla erecta*	I	0%
Deschampsia cespitosa	II	0%		*Rumex acetosa*	I	0%
Holcus mollis	II	0%		*Rumex sanguineus*	I	0%
Pteridium aquilinum	II	0%		*Stachys sylvatica*	I	0%
Solanum dulcamara	II	0%		*Teucrium scorodonia*	I	0%
Senecio jacobaea	I	29%		*Agrostis stolonifera*	-	35%
Epilobium montanum	I	12%		*Betula pendula*	-	18%
Calystigia sepium	I	6%		*Lolium perenne*	-	18%
Epilobium hirsutum	I	6%		*Taraxacum agg.*	-	18%
Poa pratensis	I	6%		*Artemisia vulgaris*	-	12%
Rumex obtusifolius	I	6%		*Buddleja davidii*	-	12%
Sambucus nigra	I	6%		*Epilobium ciliatum*	-	12%
Silene dioica	I	6%		*Hypericum perforatum*	-	12%
Achillea millefolium	I	0%		*Lactuca serriola*	-	12%
Anthoxanthum odoratum	I	0%		*Salix caprea*	-	12%
Anthriscus sylvestris	I	0%				
Athyrium felix-femina	I	0%				
Cerastium fontanum	I	0%				
Cirsium palustre	I	0%				
Circaea lutetiana	I	0%				
Digitalis purpurea	I	0%				

SH9 Artemisia vulgaris-Urtica dioica community

Species	Cover Score	Quad (17)		Species	Cover Score	Quad (17)
Artemisia vulgaris	V	100%		*Poa trivialis*	II	0%
Rumex obtusifolius	IV	68%		*Plantago major*	I	16%
Urtica dioica	IV	26%		*Lolium perenne*	I	11%
Holcus lanatus	III	63%		*Galium aparine*	*	16%
Cirsium arvense	III	32%		*Hordeum murinum*	*	16%
Artemisia absinthium	III	26%		*Stachys sylvatica*	*	16%
Elytrigia repens	III	16%		*Trifolium pratense*	*	16%
Anisantha sterilis	III	5%		*Chamerion angustifolium*	*	11%
Agrostis stolonifera	II	58%		*Crepis capillaris*	*	11%
Rubus fruticosus agg.	II	42%		*Festuca rubra*	*	11%
Plantago lanceolata	II	37%		*Geranium robertianum*	*	11%
Taraxacum	II	37%		*Lactuca serriola*	*	11%
Trifolium repens	II	32%		*Ranunculus repens*	*	11%
Calystegia sepium	II	26%		*Tripleurospermum inodorum*	*	11%
Arrhenatherum elatius	II	21%		*Acer pseudoplatanus*	*	5%
Medicago lupulina	II	16%		*Agrostis capillaris*	*	5%
Reseda luteola	II	16%		*Anthriscus sylvestris*	*	5%
Cirsium vulgare	II	11%		*Conium maculatum*	*	5%
Crepis vesicaria	II	11%		*Convolvulus arvensis*	*	5%
Epilobium hirsutum	II	11%		*Heracleum sphondylium*	*	5%
Epilobium montanum	II	11%		*Papaver dubium*	*	5%
Tussilago farfara	II	11%		*Rumex crispus*	*	5%
Malva sylvestris	II	5%		*Senecio jacobaea*	*	5%
Senecio squalidus	II	5%		*Vicia sativa*	*	5%
Arctium minus	II	0%		*Lotus corniculatus*	-	26%
Dactylis glomerata	II	0%		*Epilobium ciliatum*	-	11%
Lamium album	II	0%				

SH10 Rumicietum obtusifolii

Species	Cover Score	Quad (17)	Species	Cover Score	Quad (17)
Rumex obtusifolius	V	100%	*Vicia sativa*	II	0%
Holcus lanatus	IV	77%	*Arctium minus*	I	0%
Cirsium vulgare	IV	54%	*Ballota nigra*	I	0%
Urtica dioica	IV	15%	*Poa pratensis sens.lat.*	*	25%
Trifolium repens	III	62%	*Epilobium hirsutum*	*	23%
Cirsium arvense	III	31%	*Galium aparine*	*	23%
Elytrigia repens	III	23%	*Anisantha sterilis*	*	8%
Ranunculus repens	III	23%	*Chamerion angustifolium*	*	8%
Poa trivialis	III	32%	*Dactylis glomerata*	*	8%
Lolium perenne	II	54%	*Festuca rubra*	*	8%
Agrostis stolonifera	II	46%	*Lactuca serriola*	*	8%
Arrhenatherum elatius	II	23%	*Lathyrus pratensis*	*	8%
Epilobium montanum	II	23%	*Medicago lupulina*	*	8%
Trifolium pratense	II	23%	*Plantago major*	*	8%
Senecio squalidus	II	15%	*Rubus fruticosus agg.*	*	8%
Sonchus asper	II	15%	*Senecio jacobaea*	*	8%
Tussilago farfara	II	15%	*Taraxacum agg.*	-	38%
Artemisia vulgaris	II	8%	*Agrostis capillaris*	-	15%
Calystegia sepium	II	8%	*Crepis vesicaria*	-	15%
Heracleum sphondylium	II	0%	*Phleum pratense*	-	15%
Lamium album	II	0%	*Sonchus oleraceus*	-	15%

SH11 Melilotetum albae-officinalis

Species	Cover Score	Quad (17)	Species	Cover Score	Quad (17)
Melilotus albus	V	30%	*Elytrigia repens*	II	0%
Holcus lanatus	IV	78%	*Sonchus asper*	II	0%
Medicago lupulina	IV	78%	*Cerastium fontanum*	*	22%
Trifolium repens	IV	65%	*Festuca rubra*	*	22%
Agrostis stolonifera	III	87%	*Linaria vulgaris*	*	13%
Trifolium pratense	III	78%	*Melilotus altissimus*	*	13%
Melilotus officinalis	III	70%	*Rumex obtusifolius*	*	13%
Plantago lanceolata	III	48%	*Senecio jacobaea*	*	13%
Tripleurospermum inodorum	III	35%	*Solidago canadensis*	*	13%
Cirsium vulgare	III	30%	*Arrhenatherum elatius*	*	9%
Cirsium arvense	III	26%	*Chamerion angustifolium*	*	9%
Plantago major	III	9%	*Dactylis glomerata*	*	9%
Daucus carota	III	0%	*Poa pratensis sens.lat.*	*	9%
Lolium perenne	II	74%	*Rubus fruticosus agg.*	*	9%
Tussilago farfara	II	35%	*Achillea millefolium*	*	4%
Vulpia myuros	II	26%	*Equisetum arvense*	*	4%
Artemisia vulgaris	II	22%	*Lotus corniculatus*	*	4%
Crepis vesicaria	II	22%	*Poa annua*	*	4%
Poa trivialis	II	13%	*Polygonum aviculare*	*	4%
Vicia sativa	II	13%	*Ranunculus repens*	*	4%
Leucanthemum vulgare	II	4%	*Senecio squalidus*	*	4%
Sonchus oleraceous	II	4%	*Vicia cracca*	*	4%
Artemisia absinthium	II	0%	*Taraxacum*	-	26%

SH17 Holcus lanatus transition community

Species	Cover Score	Quad (17)		Species	Cover Score	Quad (17)
Holcus lanatus	V	97%		*Ranunculus repens*	*	14%
Medicago lupulina	IV	78%		*Plantago major*	*	13%
Trifolium repens	IV	72%		*Epilobium hirsutum*	*	9%
Plantago lanceolata	IV	57%		*Poa trivialis*	*	9%
Cirsium arvense	III	38%		*Sonchus oleraceus*	*	8%
Artemisia vulgaris	III	37%		*Linaria vulgaris*	*	7%
Vicia sativa	III	23%		*Hieracium*	*	5%
Dactylis glomerata	III	19%		*Buddleja davidii*	*	4%
Artemisia absinthium	III	8%		*Equisetum arvense*	*	4%
Chamerion angustifolium	III	5%		*Galium aparine*	*	4%
Agrostis stolonifera	II	67%		*Senecio squalidus*	*	4%
Lolium perenne	II	51%		*Urtica dioica*	*	4%
Trifolium pratense	II	44%		*Acer pseudoplatanus*	*	3%
Arrhenatherum elatius	II	28%		*Centaurea nigra*	*	3%
Cerastium fontanum	II	28%		*Cytisus scoparius*	*	3%
Epilobium montanum	II	22%		*Heracleum sphondylium*	*	3%
Festuca rubra	II	19%		*Lactuca serriola*	*	3%
Tussilago farfara	II	19%		*Lathyrus pratensis*	*	3%
Senecio jacobaea	II	18%		*Reseda luteola*	*	3%
Cirsium vulgare	II	15%		*Solidago canadensis*	*	3%
Rumex obtusifolius	II	14%		*Leucanthemum vulgare*	*	2%
Achillea millefolium	II	9%		*Melilotus albus*	*	2%
Tripleurospermum inodorum	II	8%		*Poa annua*	*	2%
Vulpia myuros	II	6%		*Tanacetum vulgare*	*	2%
Anisantha sterilis	II	3%		*Vicia hirsuta*	*	2%
Taraxacum	I	39%		*Conyza canadensis*	*	1%
Lotus corniculatus	I	9%		*Daucus carota*	*	1%
Melilotus officinalis	I	8%		*Hordeum murinum*	*	1%
Hypochaeris radicata	I	7%		*Juncus effusus*	*	1%
Elytrigia repens	I	5%		*Juncus inflexus*	*	1%
Geranium dissectum	I	3%		*Odontites vernus*	*	1%
Potentilla reptans	I	3%		*Quercus robur*	*	1%
Rubus fruticosus agg.	*	24%		*Rumex acetosella*	*	1%
Agrostis capillaris	*	21%		*Sonchus arvensis*	*	1%
Sonchus asper	*	16%		*Crepis vesicaria*	-	31%
Poa pratensis sens.lat.	*	15%				

W24 Rubus fruticosus-Holcus lanatus understorey

Species	Cover Score	Quad (17)		Species	Cover Score	Quad (17)
Rubus fruticosus agg.	V	98%		*Anisantha sterilis*	I	2%
Holcus lanatus	IV	84%		*Anthoxanthum odoratum*	I	2%
Arrhenatherum elatius	III	51%		*Centaurea nigra*	I	2%
Dactylis glomerata	III	32%		*Hypochaeris radicata*	I	2%
Galium aparine	III	31%		*Quercus robur*	I	2%
Urtica dioica	III	23%		*Sambucus nigra*	I	2%
Heracleum sphondylium	III	6%		*Silene dioica*	I	2%
Agrostis stolonifera	II	65%		*Vicia sepium*	I	2%
Chamerion angustifolium	II	32%		*Corylus avellana*	I	1%
Cirsium arvense	II	31%		*Trisetum flavescens*	I	1%
Festuca rubra	II	30%		*Alliaria petiolata*	I	0%
Taraxacum	II	22%		*Arum maculatum*	I	0%
Poa trivialis	II	14%		*Betula pubescens*	I	0%
Achillea millefolium	II	11%		*Cirsium palustre*	I	0%
Cirsium vulgare	II	11%		*Deschampsia cespitosa*	I	0%
Crataegus monogyna	II	6%		*Digitalis purpurea*	I	0%
Geum urbanum	II	6%		*Elytrigia repens*	I	0%
Anthriscus sylvestris	II	5%		*Equisetum arvense*	I	0%
Geranium robertianum	II	5%		*Fagus sylvatica*	I	0%
Hedera helix	II	4%		*Filipendula ulmaria*	I	0%
Glechoma hederacea	II	2%		*Fragaria vesca*	I	0%
Cruciata lavipes	II	0%		*Lapsana communis*	I	0%
Senecio jacobaea	I	18%		*Mercurialis perennis*	I	0%
Agrostis capillaris	I	16%		*Phalaris arundinacea*	I	0%
Lolium perenne	I	16%		*Phragmites australis*	I	0%
Epilobium montanum	I	14%		*Potentilla sterilis*	I	0%
Poa pratensis sens.lat.	I	13%		*Prunus spinosa*	I	0%
Ranunculus acris	I	11%		*Ranunculus ficaria*	I	0%
Epilobium hirsutum	I	10%		*Rumex sanguineus*	I	0%
Vicia sativa	I	10%		*Solanum dolcamara*	I	0%
Trifolium repens	I	9%		*Solidago virgaurea*	I	0%
Acer pseudoplatanus	I	8%		*Stellaria holostea*	I	0%
Fraxinus excelsior	I	8%		*Veronica chamaedrys*	I	0%
Ranunculus repens	I	5%		*Viola riviniana*	I	0%
Lotus corniculatus	I	4%		*Plantago lanceolata*	-	18%
Stachys sylvatica	I	4%		*Rumex obtusifolius*	-	17%
Ulex europaeus	I	4%		*Artemisia vulgaris*	-	13%
Dryopteris filix-mas	I	3%		*Calystegia sepium*	-	13%
Lathyrus pratensis	I	3%		*Trifolium pratense*	-	12%
Prunella vulgaris	I	3%		*Crepis capillaris*	-	11%
Rosa canina agg.	I	3%				

APPENDIX IV

The age since dereliction of the 50 survey sites in the summer of 2000. OS Maps = use of a range of OS released maps; JDT = Local councils derelict land database information; Aerial photography = CityView CD Rom 1995 release; Questionnaire = see Appendix V

SITE ID NO.	SITE NAME	AGE (YRS) SINCE DERELICTION	SOURCES USED
1	Kenyon Close	13	OS Maps / Questionnaire
2	Frankley Buidling Site	3	OS Maps
3	Erdington Large	4	JDT
4	Erdington Small	4	JDT/Aerial Photography
5	Glasscutters	3	JDT/Questionnaire
6	Platts Road	5	OS Maps / Questionnaire
7	Grange Rd	5	Questionnaire
8	Damson Parkway	4	Aerial Photography
9	Ackers	7	Aerial Photography
10	Sports Centre	5	OS Maps
11	Rubery shop site	4	OS Maps / Questionnaire
12	Five Ways	20	OS Maps
13	Kings Heath pub	5	OS Maps
14	Roman Rd	4	Aerial Photography
15	Florence Road	8	OS Maps/JDT/Aerial Photography/Questionnaire
16	Blake Street	10	Questionnaire
17	Upper Reservoir Road	15	OS Maps/Aerial Photography/Questionnaire
18	Lower Reservoir Road	15	OS Maps/Aerial Photography
19	Percy Road	5	Aerial Photography/Questionnaire
20	Cradley Heath	17	Questionnaire
21	Minworth Verge	20	OS Maps/Aerial Photography
22	Minworth Sewage	20	OS Maps
23	Bentley Mill Way Small	6	OS Maps/JDT/Questionnaire
24	Bentley Mill Way Large	14	OS Maps/JDT/Questionnaire
25	Hall Green Road	10	OS Maps/JDT/Questionnaire
26	Sandy Lane	20	OS Maps/JDT/Questionnaire
27	Vincent Drive	15	JDT
28	Crest View	12	Aerial Photography/Questionnaire
29	Saltwells Bus Stop	20	OS Maps
30	Road verge near Saltwells	15	Local Nature Reserve
31	Soho Loop	8	OS Maps/JDT/Aerial Photography/Questionnaire
32	Cole Bank Road	14	OS Maps

Continued...

Appendix IV Continued...

SITE ID NO.	SITE NAME	AGE (YRS) SINCE DERELICTION	SOURCES USED
33	Samson Quarry	20	OS Maps / Questionnaire
34	Landfill	14	OS Maps / Questionnaire
35	Burberry Brickworks	14	Wildlife Trust
36	Roundhay Cole site	8	OS Maps/JDT/Aerial Photography/Questionnaire
37	Mid-Cole	20	OS Maps / Questionnaire
38	Bearwood Railway	6	JDT
39	Wilson Rd	8	OS Maps/Aerial Photography/Questionnaire
40	Ashtead Circus	7	Aerial Photography
41	Woodlands Drive	4	JDT/Aerial Photography
42	Foxyards Road	10	Questionnaire
43	Tunnel Street	10	OS Maps / Questionnaire
44	Old Park Road	10	OS Maps/JDT/Questionnaire
45	Mounts Road	2	OS Maps/JDT/Questionnaire
46	Heath Street	15	OS Maps/Aerial Photography/Questionnaire
47	Walsall	10	JDT
48	Institute Road	12	OS Maps/Aerial Photography/Questionnaire
49	Tyseley Wharf	5	Aerial Photography
50	Brownhills	20	Questionnaire

APPENDIX V

Appendix IV is a copy of the questionnaire document sent out to the general public predominantly for the purpose of ascertaining more accurately the age at which sites had initially become derelict and subsequent activities on the site. In total questionnaires were sent out to 723 households which surrounded one or more of the 30 sites for which our information was not precise or even non-existent. 208 (29%) of questionnaires were returned.

QUESTIONNAIRE

THE UNIVERSITY OF BIRMINGHAM

SITE NAME: DISUSED RAILWAY BROWNHILLS

Please tick the appropriate boxes:

1) How long have you lived here, or in this neighbourhood?

- ☐ 0 – 5 years
- ☐ 5 – 10 years
- ☐ 10 - 20 years
- ☐ More than 20 years

2) Do you know what was on this site before it became derelict?

- ☐ Houses
- ☐ Factory
- ☐ Don't know
- ☐ Electricity Sub-station
- ☐ Quarry
- ☐ Other: _____

If you have more details about the former use of this land please tell us here:

3) How long ago did this site become derelict?

- ☐ Before 1970
- ☐ 1970-1980
- ☐ 1981-1985
- ☐ 1986-1990
- ☐ 1991-1994
- ☐ 1995-1996
- ☐ 1997-1998
- ☐ Don't know

If you can be more specific about when this site became derelict, please tell us here:

e.g., What year? Or why?

4) Who uses this site now? (*You can tick more than one box)*

- ☐ Children playing
- ☐ People walking dogs
- ☐ No-one
- ☐ Other: _____

More questions over the page!...

5) Do you remember any of the following disturbances occurring on the site? (*You can tick more than one box)*

- ☐ Ground was bull-dozed
- ☐ Fly-tipping
- ☐ Mowing
- ☐ Tipping of garden waste
- ☐ Fire
- ☐ Other

If you have more details, please tell us here:

e.g., What happened? When?

6) Have you noticed any of the following wildlife on the site? Please give examples e.g. robin

- ☐ Birds _____
- ☐ Butterflies _____
- ☐ Plants _____
- ☐ Foxes / Hedgehogs / Badgers (please underline)
- ☐ Frogs / Newts / Toads (please underline)
- ☐ Other _____

And finally.....

7) In general, what would you like to see happen to wasteland sites in the West Midlands?

Please give your opinions here:

Thank you very much for taking part in our questionnaire.

Please remember that even if you can only answer one of our questions we still want to hear from you!

APPENDIX VI

Unpublished data used in Chapter 3 for leaf dry matter and specific leaf area. Used courtesy of Dr Ken Thompson, Dept. of Animal and Plant Sciences, University of Sheffield, UK

Species	Dry Matter (mg)	S.L.A. (sqrt)	Species	Dry Matter (mg)	S.L.A. (sqrt)
Acer platanoides	6.12550	4.18233	*Betula pendula*	6.13126	3.69539
Acer pseudoplatamus	5.96041	3.64824	*Brassica napus*	No data	No data
Achillea millefolium	4.42920	5.16607	*Brassica nigra*	4.06076	4.20128
Aegopodium podagraria	5.12375	4.73346	*Bromus hordeaceus*	5.23355	4.91649
Aesculus hippocastanum	No data	No data	*Bryonia dioica*	3.57671	5.85974
Aethusa cynapium	No data	No data	*Buddleja davidii*	5.68803	3.51104
Agrimonia eupatoria	5.82666	4.15155	*Buxus sempervirens*	6.37525	3.00026
Agrostis canina	5.22885	6.14884	*Calystegia sepium*	4.29087	5.94186
Agrostis capillaris	5.00881	5.54847	*Calystegia silvatica*	4.73704	4.80443
Agrostis gigantea	No data	No data	*Campanula rotundifolia*	5.05229	4.70758
Agrostemma githago	No data	No data	*Campanula trachelium*	3.98325	6.24793
Agrostis stolonifera	5.09680	5.97053	*Capsella bursa-pastoris*	3.63628	4.94493
Aira caryophyllea	No data	No data	*Cardamine hirsuta*	No data	No data
Alchemilla vulgaris agg.	No data	No data	*Carduus nutans*	3.56898	4.03422
Alliaria petiolata	4.01692	5.93435	*Carex elata*	No data	No data
Allium vineale	4.12469	4.32519	*Carex flacca*	5.64203	4.09387
Almus glutinosa	5.73991	3.81970	*Carex hirta*	5.31789	4.52947
Alopecurus pratensis	5.09182	5.22533	*Carex nigra*	No data	No data
Anagallis arvensis	3.53739	6.13572	*Carex otrubae*	5.64120	4.27366
Anisantha sterilis	4.91332	5.60648	*Carex pendula*	5.26843	4.07342
Anthemis cotula	3.27552	4.72988	*Carex spicata*	No data	No data
Anthoxanthum odoratum	6.38190	4.48406	*Castanea sativa*	6.00315	3.64767
Anthriscus sylvestris	4.55131	4.91303	*Centaurium erythraea*	4.96920	4.54485
Anthyllis vulneraria	4.11896	3.98870	*Centaurea montana*	2.81887	7.37986
Antirrhinum majus	4.05649	3.38393	*Centaurea nigra*	3.93419	5.12792
Aphanes arvensis	5.14221	4.02831	*Cerastium arvense*	No data	No data
Aquilegia vulgaris	5.42758	4.75359	*Cerastium fontanum*	3.59610	5.40302
Arabidopsis thaliana	3.21893	5.51322	*Cerastium glomeratum*	3.77724	4.50512
Arctium minus	4.21183	4.90580	*Chaenorhinum minus*	3.59528	4.50368
Armoracia rusticana	4.28357	4.33366	*Chamerion angustifolium*	4.77442	4.34785
Arrhenatherum elatius	5.54032	4.93201	*Chelidonium majus*	4.31964	6.14050
Artemisia absinthium	4.69812	4.98051	*Chenopodium album*	4.05586	4.07472
Artemisia vulgaris	5.10931	4.65571	*Chenopodium ficifolium*	4.62954	3.85531
Aster novi-belgii	4.58667	4.47210	*Chenopodium rubrum*	3.56131	4.58656
Athyrium filix-femina	5.39918	3.47678	*Cichorium intybus*	3.68700	5.61556
Atriplex patula	3.97255	4.18024	*Circaea lutetiana*	No data	No data
Atriplex prostrata	3.79489	4.36226	*Cirsium arvense*	3.63098	4.01857
Aucuba japonica	No data	No data	*Cirsium vulgare*	3.47568	3.67890
Avena fatua	4.92128	4.04076	*Clematis vitalba*	4.40683	5.56438
Ballota nigra	4.71577	4.32379	*Conium maculatum*	4.85980	5.00399
Barbarea vulgaris	4.16964	5.16388	*Conopodium majus*	4.35294	4.89485
Bellis perennis	3.36787	5.55853	*Convolvulus arvensis*	3.73074	5.77894
Berberis thunbergii	No data	No data	*Conyza canadensis*	4.19288	5.11705
Berberis vulgaris	No data	No data	*Cornus sericea*	No data	No data
Beta vulgaris	No data	No data	*Coronopus didymus*	No data	No data

Appendix VI Continued

Species	**Dry Matter (mg)**	**S.L.A. (sqrt)**	**Species**	**Dry Matter (mg)**	**S.L.A. (sqrt)**
Corylus avellana	6.30223	3.83703	*Fraxinus excelsior*	5.86182	3.57607
Cotoneaster	No data	No data	*Fuchsia magellanica*	No data	No data
Crataegus laevigata	No data	No data	*Fumaria officinalis*	3.64831	6.10061
Crataegus monogyna	6.46342	3.39412	*Galeopsis tetrahit*	4.19529	5.38955
Crepis biennis	3.45722	5.47194	*Galium aparine*	3.73775	5.27926
Crepis capillaris	3.55105	5.51949	*Galium verum*	4.83277	4.70545
Crepis paludosa	3.73708	6.08456	*Geranium dissectum*	4.88124	4.82308
Crepis vesicaria	No data	No data	*Geranium molle*	4.31782	5.15536
Crocosmia x crocosmiiflora	No data	No data	*Geranium pusillum*	4.49356	5.03506
Cymbalaria muralis	2.91764	5.41391	*Geranium robertianum*	4.30747	5.48009
Cynosurus cristatus	4.88463	5.13435	*Geum urbanum*	5.49331	4.75844
Cytisus scoparius	4.99547	4.02495	*Glechoma hederacea*	3.99876	6.28621
Dactylorhiza fuchsii	3.56904	4.84888	*Hedera helix*	5.83810	3.32448
Dactylis glomerata	4.93440	5.26589	*Heracleum mantegazzianum*	No data	No data
Daucus carota	5.01307	5.39418	*Heracleum sphondylium*	4.69087	4.57861
Deschampsia cespitosa	5.77499	3.90718	*Hieracium*	No data	No data
Deschampsia flexuosa	5.80146	3.87609	*Hippophae rhamnoides*	5.47317	3.53116
Digitalis purpurea	4.92487	3.71429	*Holcus lanatus*	4.81754	5.78740
Dipsacus fullonum	4.08726	4.62665	*Holcus mollis*	4.64865	6.37601
Dryopteris affinis	No data	No data	*Hordeum murinum*	4.35240	6.14491
Dryopteris dilatata	5.26553	4.89142	*Hordeum vulgare*	No data	No data
Dryopteris filix-mas	5.42866	4.91807	*Humulus lupulus*	5.40742	4.22070
Elytrigia repens	5.16693	4.96211	*Hypericum calycinum*	5.73662	3.38790
Epilobium ciliatum	4.04135	5.55756	*Hypericum maculatum*	5.14644	5.36470
Epilobium hirsutum	4.55721	5.29849	*Hypericum perforatum*	5.50663	4.20110
Epilobium lanceolatum	No data	No data	*Hypericum tetrapterum*	4.45563	6.00817
Epilobium montanum	4.78745	4.83822	*Hypochaeris radicata*	3.40752	4.82499
Epilobium palustre	4.01734	6.01132	*Ilex aquifolium*	5.97504	2.59302
Epilobium parviflorum	4.00720	5.64732	*Impatiens glandulifera*	3.70325	6.18626
Epilobium tetragonum	No data	No data	*Iris pseudacorus*	4.29977	4.29179
Equisetum arvense	No data	No data	*Juncus acutiflorus*	4.69047	2.98853
Erigeron acer	4.41164	4.39506	*Juncus articulatus*	4.32424	5.71527
Erodium cicutarium	No data	No data	*Juncus bufonius*	3.69421	4.13410
Eupatorium cannabinum	4.49418	5.19990	*Juncus conglomeratus*	No data	No data
Euphorbia cyparissias	No data	No data	*Juncus effusus*	No data	No data
Euphorbia helioscopia	4.43184	6.00123	*Juncus inflexus*	No data	No data
Euphrasia officinalis agg.	4.97305	3.58742	*Knautia arvensis*	4.29053	4.52953
Euphorbia peplis	4.21169	6.68954	*Laburnum anagyroides*	5.36304	4.22183
Fagus sylvatica	6.68820	3.50572	*Lactuca serriola*	4.36821	3.88474
Fallopia convolvulus	4.47408	5.07167	*Lactuca virosa*	No data	No data
Fallopia japonica	5.11470	4.08920	*Lamium album*	4.28321	5.56886
Festuca arundinacea	4.83439	4.31912	*Lamium amplexicaule*	3.67475	4.44227
Festuca gigantea	4.64796	5.21038	*Lamium purpureum*	3.82315	5.59500
Festuca ovina	6.39980	3.29920	*Lapsana communis*	3.56293	7.25502
Festuca pratensis	5.18920	4.58968	*Lathyrus latifolius*	4.69134	4.01282
Festuca rubra	5.16786	4.20742	*Lathyrus pratensis*	5.22903	4.86450
Foeniculum vulgare	No data	No data	*Lavatera arborea*	No data	No data
Fragaria vesca	6.22933	4.18256	*Leontodon autumnalis*	3.90310	5.11340

Appendix VI Continued

Species	Dry Matter (mg)	S.L.A. (sqrt)	Species	Dry Matter (mg)	S.L.A. (sqrt)
Leontodon hispidus	3.23827	5.92910	*Persicaria bistorta*	3.94671	5.69193
Leontodon saxatilis	No data	No data	*Persicaria maculosa*	4.50781	4.59391
Leucanthemum x superbum	No data	No data	*Petasites hybridus*	4.48572	4.28500
Leucanthemum vulgare	3.60507	4.70075	*Phalaris arundinacea*	5.13122	5.11405
Ligustrum ovalifolium	No data	No data	*Phalaris canariensis*	No data	No data
Ligustrum vulgare	6.11391	3.16093	*Philadelphus coronarius*	No data	No data
Linaria purpurea	4.44737	3.81687	*Phleum pratense*	4.82659	5.99395
Linaria vulgaris	4.28198	4.84184	*Phragmites australis*	5.86069	3.93372
Linum catharticum	4.58122	6.12753	*Picea abies*	No data	No data
Lolium x boucheanum	No data	No data	*Picris hieracioides*	3.89658	5.24814
Lolium multiflorum	5.15205	4.75327	*Pilosella aurantiaca*	4.13447	4.81775
Lolium perenne	4.60737	5.14282	*Pilosella officinarum*	4.33510	4.51298
Lonicera periclymenum	5.41886	3.76545	*Pimpinella saxifraga*	5.34129	3.89026
Lotus corniculatus	4.08325	5.41141	*Pinus sylvestris*	5.89238	2.48309
Lotus pedunculatus	4.70382	4.92843	*Plantago lanceolata*	3.96469	4.85188
Lunaria annua	No data	No data	*Plantago major*	3.26676	5.51842
Lupimua x regalis	No data	No data	*Plantago media*	3.81381	4.16032
Lychnis coronaria	No data	No data	*Poa annua*	4.95922	6.08677
Lycopus europaeus	No data	No data	*Poa compressa*	No data	No data
Lysimachia nummularia	4.92531	5.25546	*Poa nemoralis*	5.65034	7.06462
Lysimachia punctata	4.95625	4.68737	*Poa pratensis sens.lat.*	5.63146	4.64077
Malus domestica	No data	No data	*Poa trivialis*	4.18753	6.37374
Malus sylvestris sens.str.	No data	No data	*Polygonum aviculare*	4.55302	5.30792
Malva moschata	4.63992	4.47409	*Populus tremula*	No data	No data
Malva parviflora	No data	No data	*Potentilla anserina*	5.29873	5.24817
Malva sylvestris	No data	No data	*Potentilla erecta*	5.63592	4.65803
Matricaria discoidea	3.66991	5.35877	*Potentilla reptans*	4.93645	5.51861
Matricaria recutita	3.30904	6.40142	*Primula veris*	4.63555	4.21122
Meconopsis cambrica	4.07404	6.14218	*Prunus avium*	No data	No data
Medicago lupulina	4.42521	5.41201	*Prunus laurocerasus*	6.05149	2.62988
Medicago sativa	5.01833	4.38152	*Prunus spinosa*	5.83851	3.79295
Melilotus albus	4.26574	5.70023	*Prunella vulgaris*	3.88963	5.50163
Melilotus altissimus	No data	No data	*Pteridium aquilinum*	5.45734	4.44784
Melilotus officinalis	4.87708	3.84709	*Pulicaria dysenterica*	3.77731	5.52651
Melissa officinalis	No data	No data	*Quercus cerris*	6.63258	3.15333
Mentha arvensis	4.12813	5.45577	*Quercus petraea*	6.25214	3.67687
Mentha spicata	5.20123	3.90012	*Quercus robur*	6.23132	3.63145
Mycelis muralis	No data	No data	*Radiola linoides*	No data	No data
Myosotis arvensis	3.51253	5.16703	*Ranunculus acris*	4.37468	4.88310
Nepeta cataria	No data	No data	*Ranunculus repens*	4.14275	5.47426
Odontites vernus	4.61459	4.01845	*Raphanus raphanistrum*	3.46366	5.06539
Oenothera glazioviana	No data	No data	*Rapistrum rugosum*	No data	No data
Origanum vulgare	4.82908	4.67645	*Reseda lutea*	3.94292	4.23348
Papaver dubium	4.24653	4.48929	*Reseda luteola*	4.11210	4.37443
Papaver rhoeas	3.78011	5.49640	*Rhinanthus minor*	4.23150	4.21343
Papaver somniferum	3.73082	5.35955	*Rhus typhina*	No data	No data
Pastinaca sativa	3.90323	6.32464	*Rorippa palustris*	No data	No data
Pentaglottis sempervirens	3.44593	5.30980	*Rosa arvensis*	5.89819	4.71925

Appendix VI Continued

Species	Dry Matter (mg)	S.L.A. (sqrt)	Species	Dry Matter (mg)	S.L.A. (sqrt)
Rosa canina agg.	No data	No data	*Syringa vulgaris*	No data	No data
Rosa tomentosa	No data	No data	*Tagetes*	No data	No data
Rubus fruticosus agg.	5.96299	3.95373	*Tanacetum parthenium*	4.14871	6.02779
Rubus idaeus	No data	No data	*Tanacetum vulgare*	4.69839	4.40465
Rumex acetosa	3.20964	5.30001	*Taraxacum*	3.78550	4.93089
Rumex acetosella	3.44584	4.63241	*Taxus baccata*	6.04257	2.65631
Rumex conglomeratus	4.23651	4.69239	*Teucrium scorodonium*	6.15188	3.46116
Rumex crispus	3.24314	5.14493	*Thlaspi arvense*	4.07284	4.76883
Rumex obtusifolius	4.11096	5.24387	*Tragopogon pratensis*	4.34055	4.26931
Sagina procumbens	4.66028	4.38529	*Trifolium arvense*	5.31795	4.12047
Salix alba	No data	No data	*Trifolium campestre*	4.91668	5.49442
Salix caprea	5.86037	3.79363	*Trifolium dubium*	4.46364	5.73629
Salix fragilis	No data	No data	*Trifolium hybridum*	5.18087	4.72416
Salix viminalis	No data	No data	*Trifolium medium*	5.12875	4.67533
Sambucus nigra	5.01373	4.38612	*Trifolium pratense*	4.46131	5.28014
Sanguisorba minor	5.87672	4.12306	*Trifolium repens*	4.22614	6.23156
Saponaria officinalis	4.46729	4.83279	*Tripleurospermum inodorum*	3.97742	4.80107
Scrophularia nodosa	4.26722	5.10291	*Trisetum flavescens*	5.52970	4.47980
Sedum acre	2.53320	3.97911	*Triticum aestivum*	No data	No data
Sedum rupestre	No data	No data	*Tussilago farfara*	3.72791	4.31215
Sedum spurium	No data	No data	*Typha latifolia*	4.31055	3.80707
Senecio x baxteri	No data	No data	*Ulex europaeus*	5.46788	3.13896
Senecio cineraria	No data	No data	*Urtica dioica*	4.61087	6.05994
Senecio jacobaea	3.38108	5.23527	*Urtica urens*	4.63127	4.58387
Senecio squalidus	3.41984	4.60190	*Verbena officinalis*	No data	No data
Senecio viscosus	No data	No data	*Verbascum thapsus*	4.62254	3.73947
Senecio vulgaris	2.89948	5.07974	*Veronica arvensis*	3.62919	5.39349
Silene dioica	3.34829	5.62243	*Veronica chamaedrys*	5.44746	4.61013
Silene latifolia	3.77661	5.06666	*Veronica filiformis*	No data	No data
Silene vulgaris	4.07269	4.50335	*Viburnum lantana*	No data	No data
Sisymbrium altissimum	3.63461	4.55822	*Viburnum opulus*	5.77733	3.97561
Sisymbrium officinale	3.72607	5.12940	*Vicia cracca*	4.88373	5.08698
Solanum dulcamara	4.33547	6.11843	*Vicia hirsuta*	4.90322	5.14603
Solanum tuberosum	No data	No data	*Vicia sativa*	4.66086	4.78239
Solidago canadensis	4.70442	5.64296	*Vicia sepium*	4.30772	7.11173
Sonchus arvensis	3.72609	4.70696	*Vicia tenuifolia*	No data	No data
Sonchus asper	3.51802	4.94749	*Vinca major*	No data	No data
Sonchus oleraceus	3.44215	5.90736	*Viola arvensis*	4.08978	4.83192
Sorbus aucuparia	6.05631	3.72993	*Viola tricolor*	4.27639	4.93873
Sorbus intermedia	No data	No data	*Viola x wittrockiana*	No data	No data
Spiraea douglasii	No data	No data	*Vulpia bromoides*	5.18422	4.70284
Stachys byzantina	No data	No data	*Vulpia myuros*	5.21442	4.50434
Stachys sylvatica	4.54817	5.77956	*Zinnia elegans*	No data	No data
Stellaria graminea	4.22713	5.24924	*Cornus sericea*	No data	No data
Stellaria media	2.89624	7.34608	*Coronopus didymus*	No data	No data
Symphoricarpos albus	No data	No data			
Symphytum officinale	3.50388	5.53676			
Symphytum x uplandicum	3.50388	5.53676			

APPENDIX VII

Seed density (per m-2) of species found in soil cores at 29 survey sites of seed bank study (Chapter 4). Site ID's relate to Table 4.1. (L) designates where a lower (5-10cm depth) core was also taken.

SPECIES	1	1(L)	2	3	4	5	6	7	7(L)	8	9	10	11	11(L)
Achillea millefolium	0	0	0	0	292	0	0	11	0	0	0	35	0	0
Agrostis capillaris	0	0	0	0	847	0	94	0	0	0	0	9	0	14
Agrostis stolonifera	847	347	2030	231	2583	906	170	1802	1643	56	73	378	223	198
Alnus glutinosa	0	23	0	0	56	0	11	11	0	139	23	17	0	13
Alopecurus pratensis	0	0	45	0	0	0	0	48	25	0	0	0	0	0
Anagallis arvensis	0	0	8	0	0	0	0	0	0	0	0	0	0	0
Anthriscus sylvestris	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Aphanes arvensis	0	0	0	0	0	8	0	0	0	0	0	0	0	0
Apium nodosum	0	0	0	0	0	0	0	0	0	0	0	0	9	0
Arabidopsis thaliana	0	0	0	167	194	0	9	0	0	17	0	0	11	0
Arrhenatherum elatius	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Artemisia absinthium	14	0	0	0	0	0	20	0	0	0	0	0	0	0
Artemisia vulgaris	167	11	0	11833	56	0	106	0	94	106	0	0	234	0
Atriplex patula	125	22	23	115	42	637	0	0	0	35	0	0	0	0
Barbarea vulgaris	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Bellis perennis	0	0	0	0	0	0	0	0	0	33	0	0	0	0
Betula pendula	23	0	0	0	0	0	0	0	0	0	23	17	0	0
Fallopia convulvulus	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Brassica nigra	0	0	0	0	0	7	0	0	0	0	14	0	0	0
Brassica rapa	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Bromus hordeaceous	0	0	0	0	0	0	0	0	0	17	0	8	0	0
Buddleja davidii	0	0	0	0	0	0	119	0	0	0	0	26	0	0
Capsella bursa-pastoris	0	11	38	192	0	40	94	24	10	19	187	33	229	0
Cardamine hirsuta	931	421	0	64	14	0	341	0	0	570	11	59	52	118
Carex elata	0	0	0	0	0	0	0	145	80	0	0	0	0	0
Carex hirta	0	0	0	0	0	0	0	44	59	0	0	0	0	0
Carex otrubae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Centaurea nigra	0	0	8	0	0	0	22	11	0	0	0	0	0	0

Appendix VII Continued...(sites 1-11)

SPECIES	1	1(L)	2	3	4	5	6	7	7(L)	8	9	10	11	11(L)
Centaurium erythraea	0	0	0	0	0	0	0	11	0	0	0	0	0	0
Cerastium fontanum	0	0	159	13	0	160	896	0	0	1757	0	10	0	0
Chamerion angustifolium	0	0	8	0	0	0	0	0	0	0	58	0	9	0
Matricaria discoidea	0	0	0	0	0	0	0	0	0	19	0	0	0	0
Chenopodium album	292	130	23	13	250	59	0	0	0	0	11	297	9	12
Chen. bonus-henricus	56	0	15	282	28	97	0	17	10	0	0	0	44	0
Chenopodium polyspermum	0	0	0	0	0	24	0	0	0	0	0	0	0	0
Chenopodium rubrum	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cirsium arvense	0	0	8	77	0	7	0	0	10	0	0	8	163	12
Cirsium vulgare	56	0	23	0	0	62	0	33	12	0	0	0	0	0
Conium maculatum	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Conyza canadensis	0	0	0	38	0	0	0	0	0	35	0	9	0	0
Coronopus didymus	28	0	0	0	0	0	0	0	0	0	0	0	0	0
Crataegus monogyna	0	0	0	0	0	0	0	0	0	37	0	0	0	0
Crepis capillaris	0	0	0	590	222	0	19	0	0	2302	0	0	0	0
Cytisus scoparius	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dactylis glomerata	0	0	0	0	0	23	0	139	277	19	0	0	0	0
Digitalis purpurea	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dipsacus fullonum	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Echium vulgare	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Epilobium agg	2931	289	38	436	403	12937	571	37	35	5317	35948	729	336	305
Eupatoria cannibimum	0	0	0	0	0	0	197	0	0	0	0	8	0	0
Euphorbia helioscopa	0	0	0	0	0	0	0	0	0	0	0	0	8	0
Euphorbia peplus	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Euphrasia officinalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Festuca rubra	28	0	0	13	0	0	0	11	34	0	0	0	0	0
Fragaria vesca	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Galium aparine	0	0	0	0	0	0	0	0	0	0	0	0	8	0
Galium mollugo	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Geranium molle	0	0	0	13	0	16	0	0	0	0	10	10	0	0
Geranium robertianum	0	0	0	0	14	0	0	0	0	0	0	0	0	0
Geum urbanum	0	0	0	0	0	7	0	0	0	0	0	0	0	0
Gnaphalium uliginosum	0	0	0	13	0	25	68	0	0	0	0	0	52	0

Appendix VII Continued...(sites 1-11)

SPECIES	1	1(L)	2	3	4	5	6	7	7(L)	8	9	10	11	11(L)
Hieracium sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Holcus lanatus	417	26	970	205	403	92	0	1062	236	1052	25	2162	21	0
Holcus mollis	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hypericum perforatum	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hypochaeris radicata	0	0	0	0	14	0	12	17	0	17	95	0	0	0
Juncus bufonius	0	0	0	0	0	1552	0	0	0	0	14	0	0	0
Juncus sp.	194	412	227	0	0	0	40	10212	1770	0	0	0	216	131
Lactuca serriola	0	0	0	0	0	0	10	0	0	0	0	0	0	0
Lapsana communis	0	0	15	38	42	10	0	0	0	0	0	34	0	0
Lathyrus pratensis	0	0	0	0	0	0	11	15	10	0	0	0	0	0
Leucanthemum x superbum	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Linaria purpurea	0	0	0	0	0	0	0	0	0	0	0	36	0	0
Linaria vulgaris	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lolium perenne	0	0	91	0	208	22	12	11	54	17	0	18	9	0
Lotus corniculatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Luzula sylvestris	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Medicago lupulina	153	36	167	705	0	0	295	0	0	19	0	0	0	0
Mentha arvensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Oenothera glazioviana	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Origanum vulgare	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Papaver dubium	0	0	0	167	0	0	0	0	0	17	0	0	0	0
Persicaria maculosa	97	22	8	13	14	23	0	33	21	19	0	0	0	0
Pilosella officinarum	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Plantago lanceolata	14	0	15	0	0	0	1126	0	0	37	0	0	169	0
Plantago major	972	392	273	205	222	73	62	213	65	1048	48	53	159	0
Poa annua	28	12	652	3859	1556	293	10	67	100	1122	1624	181	107	68
Poa compressa	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Poa humilis	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Poa nemoralis	0	0	0	0	0	0	0	0	0	0	0	50	0	0
Poa pratensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Poa trivialis	556	77	76	1026	56	6	0	162	129	0	0	759	618	563
Polygonum aviculare	56	60	8	38	28	15	10	11	13	1013	0	0	33	0
Persicaria lapathifolia	0	0	0	0	0	122	0	0	0	0	0	0	0	0
Potentilla erecta	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Appendix VII Continued...(sites 1-11)

SPECIES	1	1(L)	2	3	4	5	6	7	7(L)	8	9	10	11	11(L)
Potentilla reptans	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Prunella vulgaris	0	0	0	0	0	0	0	32	0	163	0	0	0	0
Ranunculus repens	14	0	53	0	14	33	0	100	121	0	0	0	0	25
Ranununculus fluitans	0	0	0	0	0	0	0	0	0	0	0	0	8	13
Reseda luteola	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ribes sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Rosa sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Rubus fruticosus	0	0	0	0	0	0	0	0	13	0	0	0	29	26
Rumex acetosella	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Rumex obtusifolius	583	117	341	1500	42	109	119	99	22	0	0	2909	211	145
Sagina procumbens	97	11	0	26	0	137	18	0	10	27504	0	634	43	0
Sambucus nigra	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sanguisorba minor	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Saponaria sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Scleranthus annuus	0	0	8	0	0	0	0	0	0	0	0	0	0	0
Senecio jacobaea	0	0	0	154	0	8	0	11	0	56	13	0	53	116
Senecio vulgaris	0	0	30	26	83	287	141	11	0	19	531	9	0	0
Silene dioica	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sisymbrium altissimum	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sisymbrium irio	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Sisymbrium officinale	0	27	0	1551	69	0	0	0	13	19	323	25	52	65
Solanum dulcamara	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Solidago sp.	0	0	667	372	0	0	606	0	0	37	0	26	0	0
Sonchus sp.	69	56	455	167	319	1218	96	1017	194	159	791	150	29	42
Stachys arvensis	0	11	0	13	0	0	0	0	0	0	0	17	35	0
Stachys sylvatica	0	0	0	13	0	0	0	13	0	0	13	0	8	0
Stellaria media	28	0	129	90	0	0	0	17	0	304	139	72	25	0
Taraxacum agg.	69	53	8	282	28	39	101	0	12	91	53	97	69	0
Thlaspi arvense	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Trifolium dubium	0	0	68	0	0	0	22	26	10	300	0	0	0	0
Trifolium pratense	0	0	15	0	0	0	64	0	0	0	0	0	0	0
Trifolium repens	153	24	439	90	42	6	38	13	0	1281	34	114	0	0
Tripleurospermum inodorum	0	41	0	962	0	17	31	307	66	0	25	26	338	40
Trisetum flavescens	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Appendix VII Continued...(sites 1-11)

SPECIES	1	1(L)	2	3	4	5	6	7	7(L)	8	9	10	11	11(L)
Ulex europaeus	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Urtica dioica	69	12	0	1359	0	42	65	89	80	126	0	340	3746	3126
Urtica urens	0	0	0	0	0	0	0	0	0	0	0	9	0	0
Veronica arvensis	236	0	0	0	0	0	0	0	0	0	0	0	0	0
Veronica persica	0	0	0	115	0	0	0	0	0	1185	13	0	0	13
Veronica serpyllifolia	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Vicia cracca	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Vicia hirta	0	0	0	0	0	0	0	64	135	252	13	0	0	0
Vicia sativa	0	24	0	0	0	0	23	0	0	0	0	0	0	0
Vicia sepium	0	0	0	0	0	0	0	0	31	0	0	0	0	0
Viola arvensis	0	0	0	0	0	0	0	0	0	0	44	0	0	0
Viola tricolor	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Vulpia sp.	0	0	0	13	0	0	0	0	13	0	0	0	0	0

Appendix VII Continued...(sites 12-20)

SPECIES	12	13	13(L)	14	14(L)	15	16	16(L)	17	17(L)	18	19	19(L)	20
Achillea millefolium	0	0	0	69	12	23	0	0	0	0	0	0	0	12
Agrostis capillaris	0	26	74	444	83	0	0	22	232	357	0	0	0	0
Agrostis stolonifera	7999	3309	1485	1847	1940	297	9948	1219	1546	1182	719	1955	1414	1646
Alnus glutinosa	26	0	24	28	0	0	0	22	0	37	0	0	9	25
Alopecurus pratensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Anagallis arvensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Anthriscus sylvestris	0	0	0	0	0	30	0	0	0	0	0	0	0	0
Aphanes arvensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Apium nodosum	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Arabidopsis thaliana	0	0	0	0	12	0	17	0	0	50	7	44	25	12
Arrhenatherum elatius	0	59	0	0	0	0	0	0	0	0	40	0	0	0
Artemisia absinthium	0	0	0	0	0	0	286	103	101	193	32	0	0	12
Artemisia vulgaris	0	114	51	0	0	6708	1804	394	324	189	268	24	50	293
Atriplex patula	0	0	13	0	0	169	60	0	0	0	29	0	0	23
Barbarea vulgaris	0	0	0	0	0	0	0	23	0	0	0	0	0	0

Appendix VII Continued...(sites 12-20)

SPECIES	12	13	13(L)	14	14(L)	15	16	16(L)	17	17(L)	18	19	19(L)	20
Bellis perennis	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Betula pendula	0	0	0	28	0	0	0	0	0	37	0	0	0	0
Fallopia convulvulus	0	0	0	139	0	0	0	0	0	0	0	0	0	0
Brassica nigra	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Brassica rapa	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Bromus hordeaceous	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Buddleja davidii	0	0	0	139	60	0	736	22	0	27	0	0	0	0
Capsella bursa-pastoris	12	19	64	0	0	1501	12	13	87	133	37	52	0	256
Cardamine hirsuta	0	9	0	14	12	0	0	10	1087	106	77	713	103	27
Carex elata	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Carex hirta	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Carex otrubae	0	0	0	0	0	0	0	0	0	0	0	0	0	36
Centaurea nigra	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Centaurium erythraea	0	0	0	0	0	0	0	0	0	0	0	0	0	12
Cerastium fontanum	14	36	62	14	12	0	154	11	178	43	365	838	209	88
Chamerion angustifolium	0	29	0	153	48	0	0	0	74	0	57	0	0	0
Matricaria discoidea	0	0	0	0	0	0	17	0	0	0	0	0	0	0
Chenopodium album	293	56	214	28	36	23	110	85	39	43	6	0	35	0
Chenopodium bonus-henricus	40	0	0	0	0	223	15	0	11	0	0	26	0	12
Chenopodium polyspermum	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Chenopodium rubrum	0	0	0	0	0	0	438	302	0	0	0	0	0	0
Cirsium arvense	12	28	0	56	12	0	74	0	47	15	44	17	0	70
Cirsium vulgare	0	19	26	111	24	0	0	0	40	15	13	18	0	0
Conium maculatum	0	0	0	0	0	39	0	0	11	0	0	0	0	0
Conyza canadensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Coronopus didymus	0	0	0	0	0	0	0	0	0	0	0	0	0	12
Crataegus monogyna	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Crepis capillaris	0	0	0	0	0	0	67	0	11	0	0	0	0	0
Cytisus scoparius	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dactylis glomerata	36	0	0	0	0	0	0	0	0	0	18	60	0	0
Digitalis purpurea	0	0	0	28	36	0	0	0	74	93	0	0	0	0
Dipsacus fullonum	0	0	0	0	0	0	0	0	0	0	0	9	0	0

Appendix VII Continued...(sites 12-20)

SPECIES	12	13	13(L)	14	14(L)	15	16	16(L)	17	17(L)	18	19	19(L)	20
Epilobium agg	372	1227	48	1347	571	134	11419	568	1555	505	197	232	36	187
Eupatoria cannibinum	0	0	0	97	36	0	50	21	11	0	0	0	0	0
Euphorbia helioscopa	0	11	0	0	0	0	0	0	0	0	0	0	0	0
Euphorbia peplus	0	0	0	0	0	0	0	11	0	0	0	0	0	0
Euphrasia officinalis	0	0	0	0	0	0	0	0	0	0	0	0	0	12
Festuca rubra	0	11	0	0	0	0	0	0	0	0	0	0	0	25
Fragaria vesca	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Galium aparine	0	0	0	0	0	10	0	0	0	0	0	0	0	0
Galium mollugo	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Geranium molle	0	0	0	0	0	0	0	0	0	0	7	0	0	0
Geranium robertianum	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Geum urbanum	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gnaphalium uliginosum	514	102	62	0	0	0	45	22	0	0	0	0	0	0
Hieracium sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Holcus lanatus	1085	3149	96	1236	452	795	217	88	78	0	338	2794	285	188
Holcus mollis	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hypericum perforatum	0	0	0	0	0	0	0	0	0	16	13	65	37	0
Hypochaeris radicata	0	39	0	0	0	0	0	0	0	0	0	16	0	0
Juncus bufonius	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Juncus sp.	447	0	26	42	12	0	0	38	0	45	41	238	384	127
Lactuca serriola	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lapsana communis	0	0	0	0	0	0	0	0	11	0	0	0	0	96
Lathyrus pratensis	0	31	0	0	0	0	0	0	0	0	0	16	0	0
Leucanthemum x superbum	0	0	0	0	0	0	0	0	0	0	0	0	0	12
Linaria purpurea	0	0	0	0	0	0	12	0	0	0	6	0	0	0
Linaria vulgaris	0	11	0	0	0	0	214	23	0	10	148	0	0	35
Lolium perenne	52	0	0	14	0	490	0	0	0	0	7	120	35	28
Lotus corniculatus	0	0	0	0	0	0	0	0	14	10	0	0	0	0
Luzula sylvestris	0	0	0	0	83	0	0	0	0	0	0	0	0	0
Medicago lupulina	0	568	103	0	0	0	96	0	202	37	124	359	18	86
Mentha arvensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Oenothera glazioviana	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Origanum vulgare	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Appendix VII Continued...(sites 12-20)

SPECIES	12	13	13(L)	14	14(L)	15	16	16(L)	17	17(L)	18	19	19(L)	20
Persicaria maculosa	0	0	0	0	0	0	0	21	25	0	0	138	25	0
Pilosella officinarum	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Plantago lanceolata	12	11	0	0	0	13	185	0	188	0	450	1139	95	332
Plantago major	0	9	0	42	214	0	12	13	265	120	27	0	0	120
Poa anmua	24	70	103	0	24	2239	147	10	108	148	53	58	21	80
Poa compressa	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Poa humilis	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Poa nemoralis	0	0	0	42	36	0	0	0	0	0	0	0	0	0
Poa pratensis	0	0	0	0	0	0	0	0	0	0	14	0	0	0
Poa trivialis	0	70	13	208	190	10	0	0	13	12	169	278	65	12
Polygonum aviculare	12	0	0	0	0	33	57	11	25	75	0	16	17	0
Persicaria lapathifolia	0	10	0	0	0	0	0	0	0	0	0	0	0	0
Potentilla erecta	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Potentilla reptans	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Prunella vulgaris	0	0	0	0	0	0	0	0	11	0	0	0	0	0
Ranunculus repens	0	0	0	486	71	0	0	0	0	0	0	0	0	16
Ranunnculus fluitans	0	0	0	0	0	0	0	66	0	0	0	0	0	0
Reseda luteola	0	0	0	0	0	0	91	56	22	0	0	0	0	0
Ribes sp.	0	0	0	0	12	0	0	0	0	0	0	0	0	0
Rosa sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Rubus fruticosus	154	0	0	14	0	0	0	0	14	0	0	0	0	0
Rumex acetosella	90	20	105	0	0	0	0	0	0	10	0	0	0	0
Rumex obtusifolius	267	65	24	0	24	0	176	11	13	0	45	133	71	107
Sagina procumbens	24	11	13	42	131	13	0	0	221	382	5	623	102	226
Sambucus nigra	0	0	0	14	0	0	0	0	0	0	0	0	0	0
Sanguisorba minor	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Saponaria sp.	0	0	0	0	0	0	0	0	11	0	0	0	0	0
Scleranthus annuus	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Senecio jacobaea	0	0	0	319	0	0	17	0	0	0	47	0	0	24
Senecio vulgaris	0	0	11	0	0	36	43	11	0	0	6	0	0	0
Silene dioica	0	0	0	0	0	0	0	0	0	0	12	0	0	0
Sisymbrium altissimum	0	0	0	0	0	0	0	0	0	55	0	0	0	12
Sisymbrium irio	0	0	0	0	0	23	0	0	0	11	0	0	0	0

Appendix VII Continued...(sites 12-20)

SPECIES	12	13	13(L)	14	14(L)	15	16	16(L)	17	17(L)	18	19	19(L)	20
Solanum dulcamara	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Solidago sp.	0	0	0	0	0	0	0	0	0	0	470	9	0	15674
Sonchus sp.	89	87	11	42	24	80	175	21	99	70	254	75	8	0
Stachys arvensis	0	0	0	0	0	38	0	11	41	0	0	0	0	0
Stachys sylvatica	0	0	0	0	0	0	0	0	0	0	7	0	0	0
Stellaria media	13	50	38	0	0	528	45	25	25	118	0	0	0	12
Taraxacum agg.	0	9	0	333	60	10	47	0	47	21	67	171	8	23
Thlaspi arvense	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Trifolium dubium	0	10	0	0	0	0	0	0	33	0	72	681	59	24
Trifolium pratense	13	0	0	0	0	0	17	0	0	0	20	112	15	59
Trifolium repens	236	185	103	69	12	0	17	0	69	81	36	404	0	396
Tripleurospermum inodorum	74	19	13	0	0	952	532	170	201	68	36	8	8	407
Trisetum flavescens	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ulex europaeus	25	0	0	0	0	0	17	0	0	0	0	0	0	0
Urtica dioica	26	9	0	69	24	346	651	227	566	676	316	74	89	48
Urtica urens	0	0	0	0	0	0	0	0	11	0	0	0	0	0
Veronica arvensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Veronica persica	0	0	0	14	24	0	0	0	0	0	0	0	0	0
Veronica serpyllifolia	0	0	0	0	0	0	0	0	0	59	0	0	0	0
Vicia cracca	0	0	0	0	0	0	0	0	0	0	0	8	0	0
Vicia hirta	0	38	0	28	0	0	0	0	0	0	0	0	0	77
Vicia sativa	79	10	24	0	0	0	0	0	0	0	32	0	0	52
Vicia sepium	0	9	0	0	0	0	0	0	0	0	0	0	0	0
Viola arvensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Viola tricolor	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Vulpia sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Appendix VII Continued...(sites 21-29)

SPECIES	21	22	23	23(L)	24	25	26	27	27(L)	28	28(L)	29
Achillea millefolium	0	0	106	0	28	0	90	0	0	0	0	44
Agrostis capillaris	267	0	0	0	9	49	90	0	0	1139	198	98
Agrostis stolonifera	2165	876	3238	1157	773	3876	321	482	176	542	42	505
Almus glutinosa	10	12	41	44	18	41	13	0	0	14	21	16
Alopecurus pratensis	0	0	0	0	0	0	0	0	0	0	0	0
Anagallis arvensis	0	0	0	0	0	0	0	0	0	0	0	0
Anthriscus sylvestris	0	0	0	0	0	0	0	0	0	0	0	0
Aphanes arvensis	0	0	0	0	0	0	0	0	0	0	0	0
Apium nodosum	0	0	0	0	0	0	0	0	0	0	0	0
Arabidopsis thaliana	0	12	0	0	93	33	0	0	0	0	0	24
Arrhenatherum elatius	0	0	14	30	0	0	0	0	0	0	0	0
Artemisia absinthium	0	0	275	170	28	0	0	0	0	0	0	0
Artemisia vulgaris	0	0	265	0	356	28	179	345	117	0	0	176
Atriplex patula	0	11	0	0	0	20	13	0	0	0	0	0
Barbarea vulgaris	0	0	0	0	0	0	0	0	0	0	0	0
Bellis perennis	0	0	0	0	0	0	0	0	0	0	0	0
Betula pendula	0	0	41	44	18	0	0	0	0	0	0	7
Fallopia convulvulus	0	0	0	0	0	0	0	0	0	0	0	0
Brassica nigra	0	0	0	0	0	0	0	0	0	0	0	0
Brassica rapa	0	0	0	0	0	0	0	0	0	0	0	0
Bromus hordeaceous	0	0	0	0	0	0	0	0	0	0	0	0
Buddleja davidii	0	0	0	0	0	0	13	0	0	0	0	0
Capsella bursa-pastoris	45	265	0	0	0	0	103	91	10	28	0	61
Cardamine hirsuta	19	70	9	11	10	10	103	10	22	0	0	121
Carex elata	0	0	0	0	0	0	0	0	0	0	0	0
Carex hirta	0	0	0	0	0	28	0	0	0	0	0	0
Carex otrubae	0	0	0	0	0	0	0	0	0	0	0	0
Centaurea nigra	0	0	79	0	0	0	0	0	0	0	0	0
Centaurium erythraea	0	0	0	0	18	0	0	0	0	0	0	0
Cerastium fontanum	222	143	34	0	264	208	0	34	0	0	0	22
Chamerion angustifolium	0	0	13	9	10	8	0	12	0	0	0	0
Matricaria discoidea	0	0	0	0	0	0	38	0	0	0	0	0
Chenopodium album	0	48	0	0	27	0	0	0	0	0	0	7

Appendix VII Continued...(sites 21-29)

SPECIES	21	22	23	23(L)	24	25	26	27	27(L)	28	28(L)	29
Chenopodium bonus-henricus	0	24	0	0	0	20	167	12	0	0	0	0
Chenopodium polyspermum	0	0	0	0	0	0	0	0	0	0	0	0
Chenopodium rubrum	0	0	0	0	0	0	0	0	0	0	0	0
Cirsium arvense	55	12	0	0	27	0	103	313	94	14	0	93
Cirsium vulgare	10	47	0	0	9	8	0	0	12	0	0	0
Conium maculatum	0	0	0	0	0	0	0	0	0	0	0	7
Conyza canadensis	0	0	0	0	0	0	0	0	0	0	0	0
Coronopus didymus	16	0	0	0	0	0	0	0	0	0	0	0
Crataegus monogyna	0	0	9	0	0	0	0	0	10	14	10	0
Crepis capillaris	9	0	0	19	361	164	0	0	0	0	0	0
Cytisus scoparius	0	12	0	0	0	0	0	0	0	0	0	0
Dactylis glomerata	73	22	0	0	0	0	13	106	0	14	31	24
Digitalis purpurea	0	0	0	0	0	0	0	0	0	0	0	7
Dipsacus fullonum	0	0	0	0	0	0	0	0	0	0	0	0
Echium vulgare	0	0	0	0	0	0	0	0	0	0	0	0
Epilobium agg	209	1379	535	198	687	566	1103	537	275	56	31	422
Eupatoria cannibinum	0	0	0	0	0	0	13	0	0	0	0	0
Euphorbia helioscopa	0	0	0	0	0	0	0	11	0	0	0	0
Euphorbia peplus	0	0	0	0	0	0	0	0	0	0	0	0
Euphrasia officinalis	0	0	0	0	0	0	0	0	0	0	0	0
Festuca rubra	0	12	0	0	211	0	38	91	0	28	0	0
Fragaria vesca	0	0	0	0	0	0	0	0	0	28	0	0
Galium aparine	0	0	0	0	0	0	192	0	0	0	0	0
Galium mollugo	0	0	0	0	0	0	51	0	0	0	0	0
Geranium molle	0	0	0	0	0	0	0	0	0	0	0	0
Geranium robertianum	0	0	0	0	0	0	0	0	0	0	0	0
Geum urbanum	0	0	0	0	0	0	0	0	0	0	0	0
Gnaphalium uliginosum	0	12	0	0	0	0	308	0	0	0	0	15
Hieracium sp.	0	0	0	0	0	28	0	0	0	0	10	0
Holcus lanatus	2631	1216	301	90	584	197	3179	935	181	389	31	607
Holcus mollis	0	0	0	0	0	0	0	112	0	0	0	0
Hypericum perforatum	0	47	0	0	0	0	13	12	0	0	0	199
Hypochaeris radicata	45	11	13	31	0	0	0	0	0	97	31	0
Juncus bufonius	0	0	0	0	0	0	0	0	0	0	0	0

Appendix VII Continued...(sites 21-29)

SPECIES	21	22	23	23(L)	24	25	26	27	27(L)	28	28(L)	29
Juncus sp.	75	1132	19	0	753	121	2603	20	32	14	0	781
Lactuca serriola	0	0	0	0	0	0	0	0	0	0	0	0
Lapsana communis	0	190	0	9	0	0	13	0	0	0	0	0
Lathyrus pratensis	0	79	0	0	0	0	13	35	0	28	0	31
Leucanthemum x superbum	0	0	14	0	0	0	141	0	0	0	0	0
Linaria purpurea	0	0	0	0	0	0	0	0	0	0	0	0
Linaria vulgaris	0	0	47	0	0	0	0	12	21	0	0	0
Lolium perenne	19	24	38	0	70	18	13	23	10	0	0	39
Lotus corniculatus	0	0	0	0	0	0	13	0	0	0	0	0
Luzula sylvestris	0	36	9	0	0	0	0	0	0	0	0	0
Medicago lupulina	0	36	0	0	334	160	0	161	21	42	10	7
Mentha arvensis	0	0	0	0	0	0	0	22	0	0	0	0
Oenothera glazioviana	0	0	0	0	0	0	0	0	0	0	0	7
Origanum vulgare	0	220	0	0	0	0	0	0	0	0	0	0
Papaver dubium	0	35	0	0	0	0	0	0	0	0	0	0
Persicaria maculosa	0	0	0	0	0	0	0	0	0	0	0	7
Pilosella officinarum	0	0	11	0	0	0	0	0	0	0	0	0
Plantago lanceolata	560	94	164	9	37	716	1308	760	52	0	0	361
Plantago major	87	83	11	0	531	207	38	9	43	0	0	142
Poa annua	81	152	11	0	290	19	1179	45	43	42	21	31
Poa compressa	0	0	0	35	0	0	0	0	0	69	302	0
Poa humilis	0	0	0	0	0	0	0	0	74	0	0	0
Poa nemoralis	0	0	0	0	0	0	0	0	0	42	10	0
Poa pratensis	0	0	0	0	0	0	0	12	64	69	31	0
Poa trivialis	10	71	9	0	63	39	487	554	258	28	0	575
Polygonum aviculare	0	12	0	0	18	62	77	0	10	0	0	47
Persicaria lapathifolia	0	0	0	0	0	0	0	0	0	0	0	0
Potentilla erecta	0	0	0	0	0	0	13	0	0	0	0	0
Potentilla reptans	0	23	0	0	0	11	26	12	22	0	0	0
Prunella vulgaris	0	0	0	0	0	0	0	0	0	0	0	0
Ranunculus repens	89	80	0	0	247	95	0	0	0	0	0	0
Ranunnculus fluitans	0	0	0	0	0	0	0	0	0	0	0	0
Reseda luteola	0	0	0	0	0	0	0	0	0	0	0	0
Ribes sp.	10	0	0	0	0	0	0	0	0	0	0	0

Appendix VII Continued...(sites 21-29)

SPECIES	21	22	23	23(L)	24	25	26	27	27(L)	28	28(L)	29
Rosa sp.	0	0	0	0	0	0	0	0	0	0	0	16
Rubus fruticosus	0	81	0	0	0	0	0	0	11	0	0	30
Rumex acetosella	0	0	344	326	28	0	0	0	0	14	0	0
Rumex obtusifolius	1631	352	244	0	141	220	1833	88	163	14	21	1090
Sagina procumbens	213	2704	0	0	1433	11	526	0	10	0	0	1047
Sambucus nigra	0	0	0	0	0	0	0	0	0	0	0	0
Sanguisorba minor	0	0	0	10	0	0	0	0	0	0	0	0
Saponaria sp.	0	0	0	0	0	0	0	0	0	0	0	0
Scleranthus annuus	0	0	0	0	0	0	0	0	0	0	0	0
Senecio jacobaea	183	118	0	0	9	10	179	0	0	0	0	127
Senecio vulgaris	47	0	87	0	0	0	26	0	0	0	0	16
Silene dioica	0	0	0	0	0	0	0	0	0	0	0	0
Sisymbrium altissimum	0	0	0	0	0	0	0	0	0	0	0	0
Sisymbrium irio	0	0	0	0	0	0	0	0	0	0	0	0
Sisymbrium officinale	41	0	0	10	19	0	13	21	0	0	0	39
Solanum dulcamara	0	0	11	0	0	0	0	0	0	0	0	0
Solidago sp.	0	12	0	0	0	394	0	46	10	0	0	166
Sonchus sp.	174	0	253	0	222	57	77	53	0	0	31	68
Stachys arvensis	0	24	0	0	9	0	0	12	0	0	0	0
Stachys sylvatica	0	48	0	0	0	0	0	130	105	0	0	0
Stellaria media	16	0	0	0	19	0	0	12	10	0	0	71
Taraxacum agg.	387	70	143	56	28	9	0	81	10	0	0	37
Thlaspi arvense	0	0	0	0	10	0	0	0	0	0	0	0
Trifolium dubium	35	71	0	0	85	0	77	0	0	28	10	0
Trifolium pratense	0	12	0	0	71	375	0	33	10	0	0	15
Trifolium repens	17	129	37	0	37	43	372	9	10	0	0	136
Tripleurospermum inodorum	19	92	0	30	38	40	167	0	32	0	10	30
Trisetum flavescens	0	24	0	0	0	0	0	0	0	0	0	0
Ulex europaeus	0	0	0	0	0	0	0	0	0	0	0	0
Urtica dioica	611	106	0	0	153	67	256	239	181	42	0	95
Urtica urens	0	0	0	0	9	0	0	0	0	0	0	0
Veronica arvensis	0	0	125	0	0	0	0	0	0	0	0	0
Veronica persica	0	210	0	0	9	0	0	0	0	0	0	0
Veronica serpyllifolia	0	0	0	0	0	0	0	0	0	0	0	0

Appendix VII Continued...(sites 21-29)

SPECIES	21	22	23	23(L)	24	25	26	27	27(L)	28	28(L)	29
Vicia cracca	0	0	0	0	0	0	0	0	0	0	0	0
Vicia hirta	27	271	0	0	0	0	0	46	0	83	0	55
Vicia sativa	19	24	0	0	0	0	38	0	0	14	21	0
Vicia sepium	0	12	0	0	10	0	0	0	0	0	0	15
Viola arvensis	0	22	0	0	0	0	0	0	0	0	0	0
Viola tricolor	0	0	0	0	0	0	0	21	0	0	0	0
Vulpia sp.	0	0	0	0	0	0	0	0	10	0	0	0

APPENDIX VIII

Ecorecord data on the species records for the Birmingham and Black Country region.

Ecorecord rarity index is calculated from number of km2 from which a species is recorded. Over 682 = Abundant (1); 501-682 = Very Common (2); 215-500 = Common (3); 86-214 = Frequent (4); 31-85 = Uncommon (5); 7-30 = Rare (6); 1-6 = Very Rare (7); New Records treated as Very Rare (7).

Species	Total Records	1km Squares	Species	Total Records	1km Squares
Acer platanoides	159	121	*Bellis perennis*	719	348
Acer pseudoplatanus	2755	557	*Berberis thunbergii*	3	3
Achillea millefolium	1098	376	*Berberis vulgaris*	2	No data
Aegopodium podagraria	349	196	*Beta vulgaris*	1	No data
Aesculus hippocastanum	608	No data	*Betula pendula*	691	325
Aethusa cynapium	96	No data	*Brassica napus*	84	No data
Agrimonia eupatoria	12	8	*Brassica nigra*	3	2
Agrostemma githago	10	8	*Brassica rapa*	42	36
Agrostis canina	61	32	*Bromus hordeaceous*	176	122
Agrostis capillaris	1496	435	*Bryonia dioica*	216	119
Agrostis gigantea	99	69	*Buddleja davidii*	204	138
Agrostis stolonifera	868	342	*Buxus sempervirens*	13	12
Aira caryophyllea	29	No data	*Calystegia sepium*	947	379
Alchemilla vulgaris	48	32	*Calystegia silvatica*	116	91
Alliaria petiolata	745	328	*Campanula rotundifolia*	67	32
Allium vineale	9	8	*Campanula trachelium*	25	20
Alnus glutinosus	1259	388	*Capsella bursa-pastoris*	546	300
Alopecurus pratensis	610	277	*Cardamine hirsuta*	171	No data
Anagallis arvensis	47	41	*Carduus crispus*	38	18
Anisanthus sterilis	286	178	*Carduus nutans*	55	44
Anthemis cotula	7	5	*Carex elata*	0	0
Anthoxanthum odoratum	534	226	*Carex flacca*	118	64
Anthriscus sylvestris	1703	475	*Carex hirta*	291	155
Anthyllis vulneraria	38	28	*Carex nigra*	163	71
Antirrhinum majus	9	8	*Carex otrubae*	411	165
Aphanes arvensis	5	4	*Carex pendula*	104	51
Aquilegia vulgaris	17	17	*Carex spicata*	7	5
Arabidopsis thaliana	91	67	*Castanea sativa*	173	86
Arctium minus	218	139	*Centaurea montana*	11	11
Armoracia rusticana	478	250	*Centaurea nigra*	892	306
Arrhenatherum elatius	2183	518	*Centaurium erythraea*	165	72
Artemisia absinthium	1136	333	*Cerastium arvense*	2	1
Artemisia vulgaris	1520	444	*Cerastium fontanum*	805	325
Aster novi-belgii	124	87	*Cerastium glomeratum*	24	17
Athyrium filix-femina	133	82	*Chaenorrhinum minus*	12	7
Atriplex patula	96	73	*Chamerion angustifolium*	2870	563
Atriplex prostata	52	40	*Chelidonium majus*	128	78
Aucuba japonica	1	1	*Chenopodium album*	229	154
Avena fatua	25	23	*Chenopodium ficifolium*	2	0
Ballota nigra	15	12	*Chenopodium rubrum*	6	6
Barbarea vulgaris	65	48	*Cichorium intybus*	18	13

Appendix VIII Continued

Species	Total Records	1km Squares	Species	Total Records	1km Squares
Circaea lutetiana	155	No data	*Festuca arundinacea*	59	43
Cirsium arvense	1864	486	*Festuca gigantea*	261	117
Cirsium vulgare	1084	406	*Festuca ovina*	135	59
Clematis vitalba	52	29	*Festuca pratensis*	32	24
Conium maculatum	136	79	*Festuca rubra*	862	348
Conopodium majus	335	167	*Foeniculum vulgare*	72	59
Convolvulus arvensis	389	219	*Fragaria vesca*	40	27
Conyza canadensis	13	13	*Fraxinus excelsior*	2291	503
Cornus sericea	4	No data	*Fuchsia magell..*	0	No data
Coronopus didymus	19	No data	*Fumaria officinalis*	61	53
Corylus avellana	1355	385	*Galeopsis tetrahit*	200	129
Cotoneaster spp.	68	58	*Galium aparine*	1238	435
Crataegus laevigata	24	19	*Galium verum*	111	62
Crataegus monogyna	5129	592	*Geranium dissectum*	229	155
Crepis biennis	2	2	*Geranium molle*	143	102
Crepis capillaris	575	252	*Geranium pusillum*	5	5
Crepis paludosa	1	1	*Geranium robertianum*	420	237
Crepis vesicaria	74	58	*Geum urbanum*	333	161
Crocosmia x crocosmiiflora	20	19	*Glechoma hederacea*	211	114
Cymbalaria muralis	27	16	*Hedera helix*	1172	370
Cynosurus cristatus	921	284	*Heracleum mantegazzanium*	31	20
Cytisus scoparius	1011	357	*Heracleum sphondyllium*	1798	471
Dactylis glomerata	3148	562	*Hieracium sp*	0	0
Dactyllorhiza fuchsii	78	36	*Hippophae rhamnoides*	4	3
Daucus carota	103	41	*Holcus lanatus*	1759	471
Deschampsia cespitosa	1178	335	*Holcus mollis*	879	356
Deschampsia flexuosa	571	201	*Hordeum murinum*	327	204
Digitalis purpurea	1013	337	*Hordeum vulgare*	11	11
Dipsacus fullonum	163	111	*Humulus lupulus*	44	28
Dryopteris affinis	9	5	*Hypericum calycinum*	15	12
Dryopteris dilatata	511	226	*Hypericum maculatum*	117	82
Dryopteris filix-mas	794	337	*Hypericum perforatum*	272	159
Elytrigia repens	1102	387	*Hypericum tetrapterum*	64	45
Epilobium ciliatum	172	122	*Hypochaeris radicata*	819	341
Epilobium hirsutum	1696	448	*Ilex aquifolium*	1653	400
Epilobium lanceolatum	0	0	*Impatiens glandulifera*	427	163
Epilobium montanum	575	288	*Iris pseudoacorus*	563	246
Epilobium palustre	73	36	*Iris sp. (cultivate)*	6	6
Epilobium parviflorum	60	42	*Juncus acutiflorus*	144	72
Epilobium tetragonum	32	22	*Juncus articulatus*	213	98
Equisetum arvense	1050	371	*Juncus bufonius*	196	94
Erigeron acer	16	8	*Juncus conglomeratus*	243	127
Erodium cicutarium	30	No data	*Juncus effusus*	1452	397
Eupatorium cannabinum	113	47	*Juncus inflexus*	946	313
Euphorbia cyparassia	0	No data	*Knautia arvensis*	110	59
Euphorbia helioscopa	101	82	*Laburnum anagyroides*	134	104
Euphorbia peplus	108	88	*Lactuca serriola*	63	57
Euphrasia officinalis	13	9	*Lactuca virosa*	1	No data
Fagus sylvatica	807	282	*Lamium album*	786	339
Fallopia convolvulus	27	20	*Lamium amplexicaule*	5	5
Fallopia japonica	904	351	*Lamium purpureum*	145	119

Appendix VIII Continued

Species	Total Records	1km Squares	Species	Total Records	1km Squares
Lapsana communis	343	220	*Pastinaca sativa*	28	20
Lathyrus latifolius	21	19	*Pentaglottis sempervirens*	45	36
Lathyrus pratensis	698	285	*Petasites hybridus*	138	64
Lavatera arborea	7	7	*Phalaris arundinacea*	592	223
Leontodon autumnalis	476	217	*Phalaris canariensis*	19	18
Leontodon hispidus	149	89	*Philadelphus coronarius*	8	8
Leontodon saxatilis	56	33	*Phleum pratense*	159	111
Leucanthemum vulgare	491	228	*Phragmites australis*	81	49
Leucantheum x superbum	17	16	*Picea abies*	34	22
Ligustrum ovalifolium	354	219	*Picris hieracoides*	3	0
Ligustrum vulgare	63	47	*Pilosella aurantiaca*	21	14
Linaria purpurea	72	61	*Pilosella officinarum*	243	108
Linaria vulgaris	677	242	*Pimpinella saxifraga*	47	22
Linum catharticum	56	25	*Pinus sylvestris*	298	157
Lolium multiflorum	43	30	*Plantago lanceolata*	1457	436
Lolium perenne	1594	477	*Plantago major*	1122	410
Lolium x boucheanum	0	0	*Plantago media*	34	15
Lonicera periclymenum	274	153	*Poa annua*	1078	435
Lotus corniculatus	852	366	*Poa compressa*	3	No data
Lotus pedunculatus	181	104	*Poa nemoralis*	68	37
Lunaria annua	82	69	*Poa pratensis*	736	311
Lupinus x regalis	316	148	*Poa trivialis*	806	325
Lychnis coronarius	1	1	*Polygonum aviculare*	471	259
Lycopus europaeus	576	218	*Persicaria bistorta*	69	51
Lysimachia nummularia	32	26	*Persicaria maculosa*	416	237
Lysimachia punctata	63	46	*Populus tremula*	172	111
Malus domestica	147	111	*Potentilla anserina*	129	80
Malus sylvatica	201	No data	*Potentilla erecta*	166	61
Malva moschata	76	46	*Potentilla reptans*	558	277
Malva parviflora	0	0	*Primula veris*	47	33
Malva sylvestris	166	122	*Prunella vulgaris*	300	146
Matricaria discoidea	524	280	*Prunus avium*	302	187
Matricaria recutita	19	15	*Prunus laurocerasus*	212	136
Meconopsis cambrica	12	12	*Prunus spinosa*	754	260
Medicago lupulina	580	281	*Pteridium aquilinum*	1196	366
Medicago sativa	3	1	*Pulicaria dysentrica*	50	26
Melilotus albus	49	28	*Quercus cerris*	83	52
Melilotus altissima	33	No data	*Quercus petraea*	96	57
Melilotus officinalis	284	165	*Quercus robur*	1144	385
Melissa officinalis	1	1	*Radiola linoides*	1	1
Mentha arvensis	12	8	*Ranunculus acris*	910	328
Mentha spicata	21	19	*Ranunculus repens*	1818	483
Mycelis muralis	34	28	*Raphanus raphansitrum*	149	115
Myosotis arvensis	110	83	*Rapistrum rugosum*	3	No data
Nepeta cataria	0	No data	*Reseda lutea*	97	66
Odontites vernus	102	45	*Reseda luteola*	616	251
Oenothera glazioviana	8	7	*Rhinanthus minor*	118	49
Origanum vulgare	0	0	*Rhus typhina*	3	3
Papaver dubium	57	44	*Rorippa palustris*	41	35
Papaver rhoeas	223	166	*Rosa arvensis*	196	118
Papaver somniferum	59	47	*Rosa canina*	595	283

Appendix VIII Continued

Species	Total Records	1km Squares	Species	Total Records	1km Squares
Rosa sp(cultivate)	268	141	*Tagetes sp.*	0	0
Rosa tomentosa	2	2	*Tanacetum parthenium*	106	88
Rubus fruticosus	3451	588	*Tanacetum vulgare*	319	179
Rubus idaeus	486	236	*Taraxacum agg.*	1534	No data
Rumex acetosa	759	318	*Taxus baccata*	325	145
Rumex acetosella	525	239	*Teucrium scorodonia*	117	64
Rumex conglomeratus	412	205	*Thlaspi arvense*	7	6
Rumex crispus	459	250	*Tragopogon pratensis*	321	182
Rumex obtusifolius	1697	500	*Trifolium arvense*	99	58
Sagina procumbens	170	126	*Trifolium campestre*	200	113
Salix alba	124	90	*Trifolium dubium*	435	212
Salix caprea	1291	408	*Trifolium hybridum*	84	62
Salix fragilis	981	354	*Trifolium medium*	127	90
Salix repens	0	0	*Trifolium pratense*	1410	402
Salix viminalis	174	113	*Trifolium repens*	1446	450
Sambucus nigra	3406	566	*Tripleurospermum inodorum*	677	313
Sanguisorba minor	4	2	*Trisetum flavescens*	171	81
Saponaria officinalis	24	18	*Triticum aestivum*	17	16
Scrophularia nodosa	128	86	*Tussilago farfara*	1371	414
Sedum acre	91	73	*Typha latifolia*	917	275
Sedum reflexum	36	34	*Ulex europeaus*	884	296
Sedum spurium	4	4	*Urtica dioica*	3468	593
Senecio cineraria	1	No data	*Urtica urens*	10	6
Senecio jacobaea	766	322	*Verbascum thapsus*	83	63
Senecio squalidus	664	283	*Verbena officinalis*	0	0
Senecio viscosus	23	No data	*Verbena sp. (cultivate)*	0	0
Senecio vulgaris	540	293	*Veronica arvensis*	52	41
Senecio x baxterii	0	No data	*Veronica chamaedrys*	315	160
Silene dioica	516	210	*Veronica filiformis*	45	No data
Silene latifolia	350	202	*Veronica longifolia*	0	0
Silene vulgaris	213	118	*Viburnum lantana*	8	6
Sisymbrium altissimum	57	47	*Viburnum opulus*	399	207
Sisynbrium officinale	473	273	*Vicia cracca*	615	277
Solanum dulcamara	1059	377	*Vicia hirsuta*	337	195
Solanum tuberosum	16	16	*Vicia sativa*	101	60
Solidago canadensis	383	182	*Vicia sepium*	296	168
Sonchus arvensis	286	173	*Vicia temifolia*	2	1
Sonchus asper	333	199	*Vinca major*	9	No data
Sonchus oleraceous	506	278	*Viola arvensis*	81	66
Sorbus aucuparia	846	363	*Viola tricolor*	32	27
Sorbus intermedia	164	73	*Viola x wittrischiana*	2	1
Spiraea douglasii	1	1	*Vulpia bromoides*	32	23
Stachys byzantina	0	0	*Vulpia myuros*	36	27
Stachys sylvatica	739	293	*Zinnia elegans*	0	0
Stellaria graminea	185	117			
Stellaria media	733	356			
Symphoricarpos albus	248	170			
Symphytum officinale	199	128			
Symphytum x uplandicum	73	63			
Syringa vulgaris	135	105			

APPENDIX IX

Species presence/absence in the total site lists drawn up at each of the 50 survey sites (Chapters 6 and 7).

SPECIES	NO. SITES	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10	S11	S12	S13	S14	S15	S16	S17	S18	S19	S20	S21	S22	S23	S24	S25	S26	S27	S28	S29	S30	S31	S32	S33	S34	S35	S36	S37	S38	S39	S40	S41	S42	S43	S44	S45	S46	S47	S48	S49	S50
Agrostis stolonifera	50	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Chamerion angustifolium	50	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Cirsium arvense	50	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Holcus lanatus	50	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Lolium perenne	49	1	1	1	1	1		1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Plantago lanceolata	48	1	1	1	1	1	1		1	1	1	1	1	1	1	1	1		1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Taraxacum	48	1	1	1	1	1		1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1		1	1	1
Dactylis glomerata	47	1	1	1	1		1	1	1	1	1	1	1	1		1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1		1
Trifolium repens	47	1	1	1	1	1	1	1		1	1	1	1	1	1	1	1		1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1		1	1	1	1	1	1	1
Urtica dioica	47	1	1	1	1		1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1		1	1		1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Medicago lupulina	46	1	1	1	1	1	1	1	1	1	1	1	1	1		1			1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Agrostis capillaris	45	1	1	1	1		1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1		1	1	1	1	1	1	1	1	1	1		1	1		1	1	1
Arrhenatherum elatius	45	1	1	1	1			1	1	1	1	1		1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1			1	1
Cirsium vulgare	45	1	1	1	1	1	1		1	1	1	1	1	1	1		1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1		1	1	1	1	1		1	1	1		1	1
Rubus fruticosus	45			1	1		1	1		1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1		1	1	1	1	1	1
Rumex obtusifolius	45	1	1	1	1	1	1	1	1	1	1		1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1		1	1		1	1	1	1		1	1	1	1	1	1	1		1	1	1	1
Epilobium montanum	43	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1			1	1	1	1	1	1	1	1		1		1	1	1	1	1	1	1		1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Senecio jacobaea	43	1		1	1		1		1	1	1	1	1	1	1	1	1	1		1	1	1	1	1	1	1	1	1	1		1	1	1	1	1	1		1	1	1	1		1	1	1	1	1	1	1	1	1	1	1
Festuca rubra	41	1	1	1	1				1	1	1	1		1		1	1	1	1			1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Sonchus asper	41	1	1	1	1	1	1	1		1	1	1	1	1	1	1	1		1	1			1	1	1	1	1	1	1	1	1	1					1		1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Crepis capillaris	39	1	1	1	1	1	1	1		1	1		1		1	1		1	1	1	1	1	1	1	1	1	1	1	1	1		1		1	1			1		1	1	1	1	1	1	1		1		1	1
Epilobium hirsutum	39	1	1	1	1	1	1	1	1	1	1		1	1	1	1	1			1	1		1	1	1	1	1	1	1			1		1	1	1			1	1	1		1	1	1	1	1	1	1	1	1	1	1	1
Galium aparine	39	1	1	1	1		1	1	1		1	1		1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1						1	1	1	1	1	1	1	1	1	1	1	1	1		1	1	
Cerastium fontanum	38		1	1	1			1	1	1	1	1	1		1			1	1	1	1	1	1	1	1	1	1		1	1	1	1		1	1		1		1	1	1	1	1	1	1	1	1			1	1

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Plantago major	38	1	1	1	1	1	1	1	1	1	1	1	1	1	1		1			1	1	1	1	1	1	1		1		1				1	1	1		1	1	1	1	1	1	1	1	1			1	1
Trifolium pratense	38	1	1	1		1	1	1		1	1	1	1				1		1		1	1		1	1	1	1	1		1	1	1	1	1	1	1		1	1	1	1	1	1			1	1	1	1	1
Artemisia vulgaris	37	1	1	1		1	1	1		1	1	1	1	1	1	1	1	1	1			1	1	1	1	1	1	1	1		1	1			1		1	1		1		1	1	1		1	1	1		1	
Salix caprea	36	1	1	1		1	1	1			1	1	1	1	1	1				1	1	1			1	1	1			1	1	1	1	1	1	1	1		1	1	1	1	1	1	1		1		1	1	1
Vicia sativa	35		1	1	1	1	1	1		1	1	1									1	1		1	1	1	1	1		1	1	1	1	1	1		1	1	1	1	1	1		1	1	1	1	1		1	1
Tussilago farfara	34		1	1	1	1		1		1		1	1	1		1				1	1	1		1	1	1	1	1		1	1	1			1	1		1		1		1	1	1	1	1	1	1	1	1	1
Achillea millefolium	33	1	1	1	1			1	1	1	1	1	1				1				1	1	1	1	1	1		1	1	1	1	1		1	1	1	1	1		1	1	1	1	1		1				
Calystegia silvatica	33			1	1					1	1	1	1	1		1	1		1	1	1		1	1	1	1	1	1	1	1		1		1		1	1	1	1		1	1	1	1	1		1		1	
Sonchus oleraceus	32	1	1	1	1	1	1	1	1		1	1	1	1		1	1	1		1			1	1	1		1				1		1	1		1		1		1			1	1	1	1		1	1
Vicia hirsuta	32	1	1							1	1	1				1	1	1	1			1			1	1	1	1		1	1	1	1	1	1	1	1	1	1	1	1	1	1			1		1	1
Crataegus monogyna	31	1	1	1	1					1	1	1	1	1		1				1	1	1			1	1		1	1	1	1		1	1	1	1	1	1		1		1					1	1			1
Leontodon autumnalis	31			1	1	1		1	1	1	1	1	1		1			1	1			1	1	1		1	1	1	1	1	1		1					1		1	1		1	1	1	1	1			1
Poa pratensis	31	1	1	1	1		1	1	1	1	1	1	1		1	1		1	1		1	1	1				1		1	1	1		1	1			1	1			1	1		1		1	1				
Acer pseudoplatanus	30	1		1	1	1		1			1	1	1		1	1		1	1		1	1			1	1		1				1	1			1		1	1	1	1	1	1	1	1	1		1		
Hypochaeris radicata	30							1		1		1	1	1	1	1	1			1			1	1	1	1			1	1	1	1	1	1			1	1		1	1	1	1	1	1	1		1		1
Senecio squalidus	30			1		1	1	1		1	1	1	1	1		1		1		1				1		1	1		1	1	1	1							1	1	1	1	1	1	1		1	1	1	1
Betula pendula	29	1			1			1			1		1	1	1	1		1			1		1	1	1	1		1	1			1			1			1	1	1	1	1	1	1		1	1	1		1
Ranunculus repens	29	1	1	1	1		1	1	1		1	1		1	1	1				1	1			1	1	1									1				1	1	1	1	1	1	1	1	1			1	1
Poa annua	28		1	1		1	1	1		1	1		1	1		1				1			1	1	1	1		1			1				1	1	1	1			1	1		1	1	1			1		1
Poa trivialis	28	1	1	1	1		1				1	1	1	1			1											1		1		1			1		1	1		1	1	1	1	1	1	1	1	1	1	1	1
Tripleurospermum inodorum	28		1	1		1	1		1	1	1	1			1		1			1			1	1	1		1	1			1			1	1			1	1	1		1		1	1	1	1		1	1
Artemisia absinthium	27	1			1	1			1	1			1			1		1		1				1	1	1	1	1				1	1	1	1					1	1	1	1	1	1	1				1
Elytrigia repens	27	1	1	1		1	1	1			1		1	1		1	1			1		1			1	1				1		1		1			1	1	1	1	1		1	1	1	1	1	1	1
Lotus corniculatus	27		1			1			1		1	1		1			1		1					1	1	1		1	1	1	1		1	1	1	1	1		1	1	1	1	1			1	1	1		
Cytisus scoparius	26	1			1		1		1	1	1		1			1		1	1					1	1	1	1	1		1	1	1	1			1	1	1	1		1	1				1		1		
Geranium dissectum	26	1							1	1		1				1			1	1		1	1	1	1	1		1	1	1		1		1	1	1	1		1	1				1	1			1		
Anisantha sterilis	25	1		1	1		1		1					1		1	1			1		1			1	1					1	1		1	1		1	1		1			1	1	1	1	1	1				

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Buddleja davidii	25			1	1	1	1	1		1	1		1	1		1				1	1			1			1		1			1									1	1		1	1	1	1	1	1	1	
Potentilla reptans	25		1	1					1	1	1	1		1				1			1		1	1	1		1		1		1		1	1		1	1		1		1	1		1		1		1	1	1	1
Reseda luteola	25						1	1		1	1	1	1						1				1	1	1	1		1		1		1			1		1		1	1			1	1	1	1	1	1	1	1
Phleum pratense	24	1	1	1	1			1	1			1					1		1	1	1				1	1			1	1	1		1					1	1		1	1	1					1		1		
Ranunculus acris	24		1								1			1		1	1			1	1			1		1	1	1	1	1	1	1	1		1	1		1	1	1	1	1							1		
Lactuca serriola	22		1	1	1	1	1		1			1		1	1	1	1			1			1														1			1	1		1	1	1	1			1	1
Sambucus nigra	22	1					1	1						1			1						1	1	1		1		1	1								1			1	1		1	1	1		1	1	1	1
Sisymbrium officinale	21			1	1	1	1	1		1		1			1				1	1			1		1													1			1	1	1	1		1	1	1	1			
Stachys sylvatica	21	1	1		1			1			1	1									1	1			1				1				1		1		1	1			1		1	1	1					1
Centaurea nigra	20		1	1						1		1					1		1		1		1	1	1		1		1	1			1		1	1	1					1			1		1			
Equisetum arvense	20		1			1					1	1	1		1								1		1	1	1				1		1	1		1	1					1		1		1		1	1
Heracleum sphondylium	20										1	1		1	1		1	1				1		1		1	1	1		1			1	1	1		1	1	1								1			1
Hypericum perforatum	19									1			1							1	1	1	1		1	1		1		1	1					1		1		1		1		1				1		1	1
Linaria vulgaris	19												1										1	1	1		1	1			1	1		1	1			1			1	1	1	1		1		1	1
Rumex acetosa	19		1	1	1					1		1	1					1			1	1			1		1	1		1	1					1			1					1	1	1			
Solidago canadensis	19			1	1					1	1		1			1	1			1	1				1	1	1	1				1				1				1	1	1			1				
Trifolium dubium	19		1								1	1		1									1	1	1	1	1	1		1				1			1			1	1			1	1		1	1	1
Hieracium	18	1									1		1										1	1	1	1	1	1	1		1	1		1	1	1				1			1			1			1
Melilotus officinalis	18						1		1									1					1	1		1	1			1	1		1			1			1	1	1		1			1	1		1
Rosa canina agg.	18	1		1	1				1		1	1	1		1		1	1		1	1										1		1		1			1		1					1				
Conyza canadensis	17			1	1	1	1		1		1	1		1		1										1	1				1									1	1		1	1	1				
Deschampsia cespitosa	17		1	1				1			1			1						1		1	1	1	1							1		1						1			1				1	1		1
Leontodon saxatilis	17	1		1		1		1			1	1	1	1	1						1	1		1			1		1	1			1	1			1													
Polygonum aviculare	17		1	1		1	1	1	1	1	1	1					1						1															1	1		1			1							1
Prunella vulgaris	17	1	1					1	1		1			1						1									1	1	1	1		1						1	1	1				1		1	
Dipsacus fullonum	16			1									1			1							1	1	1	1	1	1		1					1	1		1		1									1	
Fraxinus excelsior	16	1				1			1	1	1	1	1				1	1			1					1		1			1						1			1							1			

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Hordeum murinum	16			1	1	1	1								1	1	1		1	1		1																1	1			1									
Leucanthemum vulgare	16		1						1	1	1		1											1	1		1			1	1			1	1		1			1							1				
Persicaria maculosa	16		1	1					1		1	1					1						1	1	1	1	1					1									1				1		1				1
Tragopogon pratensis	16											1												1	1	1	1	1		1		1	1	1			1		1		1				1	1				1
Epilobium ciliatum	15			1				1					1								1											1									1	1	1	1	1	1	1	1	1		1
Senecio vulgaris	15		1	1		1		1	1	1		1	1		1		1							1			1														1				1	1					
Armoracia rusticana	14					1					1					1		1						1	1	1		1				1		1	1		1	1			1										
Lathyrus pratensis	14		1								1	1												1		1						1	1	1		1	1	1	1	1			1				1				
Sorbus aucuparia	14	1			1	1		1	1	1			1				1			1						1								1						1		1		1							
Anthriscus sylvestris	13		1							1	1	1		1	1		1					1	1																		1			1			1	1				
Cynosurus cristatus	13		1									1	1						1		1						1			1	1				1	1	1											1		1
Geranium robertianum	13			1				1			1			1		1				1	1	1			1				1					1								1			1						
Quercus petraea	13			1											1	1		1				1						1	1			1		1		1	1	1										1			
Quercus robur	13							1	1	1	1	1	1			1	1	1		1		1																					1										1
Fallopia japonica	12									1			1				1			1		1			1	1											1		1	1	1				1						
Lapsana communis	12			1							1				1																	1									1	1		1			1	1	1	1	1
Silene latifolia	12			1				1									1		1						1	1				1	1									1		1		1	1					
Sonchus arvensis	12								1	1					1					1		1	1		1	1															1	1				1		1			
Trifolium arvense	12	1																					1	1		1			1		1	1	1								1		1		1	1				
Anthoxanthum odoratum	11		1																	1	1							1		1	1	1				1										1		1		1
Chenopodium album	11		1	1		1	1		1			1																												1	1	1			1		1			
Convolvulus arvensis	11			1		1		1																1		1	1	1															1	1	1		1					
Digitalis purpurea	11			1			1			1	1	1				1					1				1								1									1			1						
Linaria purpurea	11															1		1	1	1	1		1	1				1												1	1			1						
Papaver dubium	11					1	1									1	1			1																				1		1		1		1				1	1
Ulex europaeus	11											1						1					1		1		1		1				1	1	1	1					1									
Vicia cracca	11											1											1	1			1						1														1		1	1

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Vicia sepium	11			1																							1					1	1					1	1	1	1	1							1	1
Alopecurus pratensis	10		1	1				1				1									1								1														1				1		1	1
Dryopteris filix-mas	10			1							1		1						1		1										1		1				1					1					1			
Juncus inflexus	10		1									1														1				1		1		1			1		1							1	1		
Malus domestica	10				1							1				1					1	1			1	1														1		1			1						
Malva sylvestris	10					1														1							1										1	1		1	1	1			1						1
Tanacetum parthenium	10			1				1								1				1					1			1												1	1			1	1					
Vulpia bromoides	10					1							1							1		1		1				1		1			1		1				1										
Almus glutinosa	9										1	1		1			1		1						1							1								1		1								
Capsella bursa-pastoris	9			1			1			1								1																			1					1		1	1					1
Cotoneaster	9		1		1								1	1		1										1		1														1		1						
Geranium molle	9					1							1														1		1			1	1				1	1							1				
Trisetum flavescens	9	1		1	1										1				1																					1	1	1			1					
Aster novi-belgii	8								1			1							1																	1						1		1				1		1	
Crepis vesicaria	8					1		1											1															1							1				1	1	1			
Hedera helix	8			1				1				1	1								1																				1			1				1			
Lathyrus latifolius	8				1										1	1								1			1		1													1							1	
Oenothera glazioviana	8					1							1							1					1					1											1	1							1	
Papaver rhoeas	8			1	1															1				1			1	1						1							1						1		
Rumex crispus	8																						1					1						1	1	1		1						1	1				
Salix fragilis	8					1								1			1			1												1		1			1								1				
Trifolium campestre	8							1																			1		1					1				1				1	1		1			
Aegopodium podagraria	7															1				1		1		1																			1			1				1
Campanula trachelium	7			1									1	1							1																			1	1		1								
Centaurium erythraea	7															1									1				1		1	1			1														1
Euphorbia helioscopia	7			1				1												1					1																			1						1
Geum urbanum	7													1					1		1								1															1			1	1		

Appendix IX Continued

SPECIES	NO. SITES	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10	S11	S12	S13	S14	S15	S16	S17	S18	S19	S20	S21	S22	S23	S24	S25	S26	S27	S28	S29	S30	S31	S32	S33	S34	S35	S36	S37	S38	S39	S40	S41	S42	S43	S44	S45	S46	S47	S48	S49	S50
Lamium album	7														1		1						1																1			1	1		1						
Ligustrum ovalifolium	7			1	1													1										1		1											1								1		
Lolium multiflorum	7			1																			1		1		1											1		1					1						
Lupinus x regalis	7			1									1												1	1	1	1															1								
Myosotis arvensis	7														1					1	1	1			1															1		1									
Raphanus raphanistrum	7			1					1	1							1			1					1															1											
Vulpia myuros	7																									1														1	1			1	1				1	1
Atriplex patula	6			1				1							1																													1	1				1	
Conopodium majus	6			1					1	1														1																	1		1								
Foeniculum vulgare	6																									1		1														1	1			1			1	
Ilex aquifolium	6	1			1			1											1		1																					1									
Juncus effusus	6																															1		1	1	1		1									1		
Laburnum anagyroides	6		1											1			1									1	1																					1			
Malva moschata	6														1												1		1			1		1	1															
Matricaria discoidia	6																				1																					1		1			1		1			1
Prunus avium	6		1	1													1		1							1																					1				
Prunus spinosa	6								1	1		1					1		1																																	
Sedum reflexum	6												1					1								1			1												1							1			
Silene vulgaris	6		1	1													1								1	1																								
Solanum dulcamara	6		1				1																																	1					1	1		1	
Viola arvensis	6			1					1						1																													1					
Avena fatua	5								1								1											1																			1			
Calystegia sepium	5																												1		1									1	1					1			
Chelidonium majus	5				1	1	1																		1																1								
Festuca gigantea	5		1						1		1		1																																	1			
Festuca pratensis	5																																	1		1		1		1									
Holcus mollis	5																									1				1					1		1		1										

Appendix IX Continued

SPECIES	NO. SITES	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10	S11	S12	S13	S14	S15	S16	S17	S18	S19	S20	S21	S22	S23	S24	S25	S26	S27	S28	S29	S30	S31	S32	S33	S34	S35	S36	S37	S38	S39	S40	S41	S42	S43	S44	S45	S46	S47	S48	S49	S50
Lamium purpureum	5	1		1													1																								1						1			
Leucanthemum x superbum	5			1																						1	1													1			1						
Populus tremula	5									1	1																		1								1									1			
Pteridium aquilinum	5											1	1						1		1										1																		
Sagina procumbens	5			1										1																															1	1	1			
Stellaria media	5		1	1													1										1																						
Tanacetum vulgare	5			1															1	1						1								1													
Verbascum thapsus	5			1				1															1		1																			1					
Aethusa cynapium	4																																							1	1			1		1			
Alliaria petiolata	4									1													1		1																	1							
Atriplex prostrata	4			1			1		1																																			1					
Bellis perennis	4													1													1		1															1					
Brassica nigra	4		1									1			1																									1								
Bryonia dioica	4			1													1				1				1																							
Carex otrubae	4																					1	1					1													1								
Chenopodium rubrum	4		1				1																																							1	1			
Festuca ovina	4												1																												1			1						1
Impatiens glandulifera	4									1					1																												1		1				
Lonicera periclymenum	4			1	1														1																													1
Melilotus albus	4																									1														1				1					1	
Mentha arvensis	4																1	1																							1				1				
Mentha spicata	4																	1		1		1			1				1																			
Philadelphus coronarius	4			1													1			1										1																		
Rumex acetosella	4																																	1	1		1				1								
Salix viminalis	4	1																																1					1								1	
Silene dioica	4							1	1													1					1																				
Sisymbrium altissimum	4																					1																				1	1		1				

Appendix IX Continued

SPECIES	NO. SITES	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10	S11	S12	S13	S14	S15	S16	S17	S18	S19	S20	S21	S22	S23	S24	S25	S26	S27	S28	S29	S30	S31	S32	S33	S34	S35	S36	S37	S38	S39	S40	S41	S42	S43	S44	S45	S46	S47	S48	S49	S50
Sorbus intermedia	4																													1	1									1						1					
Stellaria graminea	4									1																	1											1		1											
Symphoricarpos albus	4		1																					1																				1			1				
Symphytum x uplandicum	4														1																								1					1				1		
Veronica chamaedrys	4																		1					1	1														1											
Acer platanoides	3	1							1								1																																		
Agrimonia eupatoria	3			1																1			1																												
Allium vineale	3																			1							1											1													
Anagallis arvensis	3							1								1																																			
Anthemis cotula	3		1																		1					1																									
Anthyllis vulneraria	3																									1				1																		1		
Arctium minus	3																								1																								1		
Aucuba japonica	3																			1																															
Bromus hordeaceus	3														1	1																																			
Carduus acanthoides	3																												1										1		1										
Carex pendula	3																			1																															
Centaurea montana	3			1							1																1																								
Chenopodium ficifolium	3																					1																													
Corylus avellana	3				1							1																																							
Crataegus laevigata	3																																																		
Cymbalaria muralis	3			1																					1																									
Daucus carota	3																																																	
Eupatorium cannabinum	3			1												1																																		
Glechoma hederacea	3										1											1																													
Hippophae rhamnoides	3	1																																																
Knautia arvensis	3																																																	
Leontodon hispidus	3																																																	

Appendix IX Continued

SPECIES	NO. SITES	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10	S11	S12	S13	S14	S15	S16	S17	S18	S19	S20	S21	S22	S23	S24	S25	S26	S27	S28	S29	S30	S31	S32	S33	S34	S35	S36	S37	S38	S39	S40	S41	S42	S43	S44	S45	S46	S47	S48	S49	S50
Lycopus europaeus	3												1	1																						1																
Lysimachia punctata	3																								1			1						1																		
Matricaria recutita	3		1	1																								1																								
Mycelis muralis	3																			1						1			1																								
Phragmites australis	3																										1																										
Picris hieracioides	3																									1																										
Pilosella aurantiaca	3																				1															1																
Pilosella officinarum	3																											1						1																		
Prunus laurocerasus	3	1														1					1																																
Pulicaria dysenterica	3																																		1			1														
Rosa arvensis	3															1												1												1												
Salix alba	3																																	1								1										
Saponaria officinalis	3																		1						1										1																
Sedum spurium	3																				1				1			1																								
Spiraea douglasii	3																		1	1														1																		
Symphytum officinale	3		1	1																																															1		
Teucrium scorodonia	3													1															1																						1		
Thlaspi arvense	3											1														1																										
Trifolium hybridum	3	1		1						1																																										
Triticum aestivum	3					1								1																																						
Veronica arvensis	3																				1																																
Viola tricolor	3																																																		
Agrostis gigantea	2																																1																		
Aquilegia vulgaris	2			1																															1																
Barbarea vulgaris	2		1																																																
Berberis thunbergii	2									1																																									
Berberis vulgaris	2																																																		

Appendix IX Continued

SPECIES	NO. SITES	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10	S11	S12	S13	S14	S15	S16	S17	S18	S19	S20	S21	S22	S23	S24	S25	S26	S27	S28	S29	S30	S31	S32	S33	S34	S35	S36	S37	S38	S39	S40	S41	S42	S43	S44	S45	S46	S47	S48	S49	S50
Brassica rapa	2			1				1																																										
Campanula rotundifolia	2				1																												1																		
Cardamine hirsuta	2																																																1		1
Carex nigra	2		1																															1																
Castanea sativa	2											1							1																																	
Cerastium arvense	2															1														1																					
Cerastium glomeratum	2																								1							1																		
Chaenorhinum minus	2											1																																							1	
Conium maculatum	2					1																																									1				
Cornus sericea	2																																																1		1	
Epilobium lanceolatum	2																										1																							1	
Epilobium palustre	2																																1															1			
Epilobium parviflorum	2																																1																1		
Epilobium tetragonum	2							1							1																																				
Euphorbia cyparissias	2																																															1		1		
Euphorbia peplis	2		1																																													1			
Fagus sylvatica	2																1																																		1		
Fragaria vesca	2																																																		1		1
Fumaria officinalis	2		1																																																
Geranium pusillum	2		1		1																																														
Hordeum vulgare	2				1																																												1		
Hypericum calycinum	2																	1	1																																	
Hypericum maculatum	2																		1							1																										
Juncus articulatus	2																																		1	1																
Juncus bufonius	2	1			1																																														
Ligustrum vulgare	2														1											1																										
Lotus pedunculatus	2																													1					1																

Appendix IX Continued

SPECIES	NO. SITES	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10	S11	S12	S13	S14	S15	S16	S17	S18	S19	S20	S21	S22	S23	S24	S25	S26	S27	S28	S29	S30	S31	S32	S33	S34	S35	S36	S37	S38	S39	S40	S41	S42	S43	S44	S45	S46	S47	S48	S49	S50
Lysimachia nummularia	2																				1				1																											
Malus sylvestris sens.str.	2																																									1							1				
Medicago sativa	2																																		1													1				
Melilotus altissimus	2																																														1			1		
Nepeta cataria	2																																														1			1		
Odontites vernus	2																										1																				1					
Origanum vulgare	2																																1							1											
Papaver somniferum	2																																	1							1										
Pentaglottis sempervirens	2																	1																									1										
Persicaria bistorta	2										1																								1																	
Phalaris arundinacea	2																								1															1												
Phalaris canariensis	2			1																																										1						
Picea abies	2	1																			1																																
Plantago media	2																														1				1																
Poa nemoralis	2																			1																									1								
Radiola linoides	2										1				1																																						
Rhinanthus minor	2																														1				1																
Rhus typhina	2		1																		1																																
Rosa sp.	2		1																																																	
Rosa tomentosa	2																				1				1																											
Rubus idaeus	2																				1																																
Rumex conglomeratus	2																			1																1																
Sanguisorba minor	2																														1					1															
Senecio viscosus	2																																														1					1
Solanum tuberosum	2														1																																1					
Syringa vulgaris	2			1																								1																							
Tagetes	2																				1																																

Appendix IX Continued

SPECIES	NO. SITES	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10	S11	S12	S13	S14	S15	S16	S17	S18	S19	S20	S21	S22	S23	S24	S25	S26	S27	S28	S29	S30	S31	S32	S33	S34	S35	S36	S37	S38	S39	S40	S41	S42	S43	S44	S45	S46	S47	S48	S49	S50
Trifolium medium	2																														1																				1
Typha latifolia	2										1																									1																
Viburnum opulus	2	1																																			1															
Aesculus hippocastanum	1																																																		1		
Agrostis canina	1																	1																																				
Agrostemma githago	1																																																			
Aira caryophyllea	1																																																			1
Alchemilla vulgaris agg.	1																									1																										
Antirrhinum majus	1			1																																															
Aphanes arvensis	1								1																																											
Arabidopsis thaliana	1		1																																																
Athyrium filix-femina	1												1																																							
Ballota nigra	1			1																																															
Beta vulgaris	1																																																	1		
Brassica napus	1																																																	1		
Buxus sempervirens	1																																																	1		
Carduus nutans	1																								1																											
Carex elata	1																																																	1		
Carex flacca	1																																																	1		
Carex hirta	1											1																																								
Carex spicata	1																																																	1		
Cichorium intybus	1																															1																			
Circaea lutetiana	1																																																		1	
Clematis vitalba	1												1																																							
Coronopus didymus	1																																																		1	
Crepis biennis	1								1																																											
Crepis paludosa	1																					1																														

Appendix IX Continued

SPECIES	NO. SITES	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10	S11	S12	S13	S14	S15	S16	S17	S18	S19	S20	S21	S22	S23	S24	S25	S26	S27	S28	S29	S30	S31	S32	S33	S34	S35	S36	S37	S38	S39	S40	S41	S42	S43	S44	S45	S46	S47	S48	S49	S50
Crocosmia x crocosmiiflora	1			1																																															
Dactylorhiza fuchsii	1																																			1																
Deschampsia flexuosa	1																																					1													
Dryopteris affinis	1											1																																							
Dryopteris dilatata	1												1																																						
Erigeron acer	1																								1																										
Erodium cicutarium	1																																																1		
Euphrasia officinalis	1																											1																						
Fallopia convolvulus	1			1																																													
Festuca arundinacea	1																																					1												
Fuchsia magellanica	1																																															1		
Galeopsis tetrahit	1																				1																													
Galium verum	1																														1																		
Heracleum mantegazzianum	1											1																																						
Humulus lupulus	1																																														1		
Hypericum tetrapterum	1																																1																
Iris pseudacorus	1																																		1														
Iris spp.	1		1																																													
Juncus acutiflorus	1																																													1		
Juncus conglomeratus	1																																1																
Lactuca virosa	1																																														1		
Lamium amplexicaule	1																					1																												
Lavatera arborea	1																			1																														
Linum catharticum	1																														1																		
Lolium x boucheanum	1																					1																												
Lunaria annua	1		1																																														
Lychnis coronaria	1																																													1		

Appendix IX Continued

SPECIES	NO. SITES	S1	S2	S3	S4	S5	S6	S7	S8	S9	S10	S11	S12	S13	S14	S15	S16	S17	S18	S19	S20	S21	S22	S23	S24	S25	S26	S27	S28	S29	S30	S31	S32	S33	S34	S35	S36	S37	S38	S39	S40	S41	S42	S43	S44	S45	S46	S47	S48	S49	S50
Malva parviflora	1															1																																				
Meconopsis cambrica	1			1																																															
Melissa officinalis	1																																	1																	
Pastinaca sativa	1																													1																				
Petasites hybridus	1																																	1																
Pimpinella saxifraga	1																																1																	
Pinus sylvestris	1																															1																		
Poa compressa	1																																																	1
Potentilla anserina	1							1																																										
Potentilla erecta	1																																				1													
Primula veris	1																													1																				
Quercus cerris	1												1																																					
Rapistrum rugosum	1																																																1	
Reseda lutea	1																												1																					
Rorippa palustris	1																1																																	
Scrophularia nodosa	1																																																
Sedum acre	1																																																
Senecio x baxteri	1																																																
Senecio cineraria	1																																																1
Stachys byzantina	1																																																
Taxus baccata	1																																																
Urtica urens	1																																																
Verbena	1																																																
Verbena officinalis	1																																																
Veronica filiformis	1																																																
Viburnum lantana	1																																																
Vicia tenuifolia	1																																																
Vinca major	1																																																
Viola x wittrockiana	1																																																
Zinnia elegans	1																																																

APPENDIX X

Data on the characteristics of the landscape surrounding each site

***(a)** Major categories of surrounding land use classified during field mapping at 100m radius and their respective percentage cover at each site in the buffer region. URB100. = All concrete / tarmac roads and non-residential development; SIM100 = similar derelict type habitat patches; MOWN100 = all mown recreational grassland and lawns; WOOD100 = Woodland types; SCRUB100 = scrub; Plant100 = planted ornamental beds; GARD100 = Residential houses with gardens. Other categories were utilised but are not displayed. Categories in **bold** were utilised for analysis in either or both Chapters 6 & 7.*

SITE ID	**SITE NAME**	AREA IN BUFFER (m^2)	**URB 100**	**SIM 100**	MOWN 100	WOOD 100	SCRUB 100	**PLANT 100**	**GARD 100**
1	Kenyan Close	43404	**21.2%**	**0.2%**	6.6%	1.2%	0.7%	**0.5%**	**56.9%**
2	Frankley	67179	**16.8%**	**4.9%**	2.1%	5.5%	26.7%	**1.4%**	**18.7%**
3	Erdington Large	56549	**26.0%**	**3.5%**	3.1%	0.0%	0.0%	**0.0%**	**57.3%**
4	Erdington Small	44009	**39.6%**	**4.7%**	4.8%	0.0%	0.0%	**0.0%**	**45.0%**
5	Glasscutters	43701	**86.6%**	**0.8%**	0.0%	0.0%	0.0%	**2.3%**	**8.3%**
6	Platts Road	50113	**62.6%**	**0.0%**	1.5%	0.0%	0.0%	**10.4%**	**20.0%**
7	Grange Rd	40355	**35.7%**	**0.0%**	0.0%	0.0%	0.0%	**3.8%**	**52.4%**
8	Solihull Arable	74878	**3.5%**	**44.5%**	3.0%	0.0%	17.2%	**1.4%**	**4.5%**
9	Ackers	43257	**32.8%**	**14.5%**	9.0%	0.0%	26.2%	**1.2%**	**0.0%**
10	Sports Centre	57630	**46.7%**	**10.4%**	19.1%	4.9%	0.0%	**2.6%**	**0.0%**
11	Rubery Shops	54498	**34.0%**	**16.1%**	24.8%	0.0%	6.2%	**1.5%**	**12.6%**
12	Five Ways	94987	**34.3%**	**11.2%**	0.8%	3.2%	0.0%	**1.1%**	**40.0%**
13	Kings Heath Pub	40670	**19.2%**	**1.2%**	26.7%	19.1%	18.5%	**0.0%**	**2.4%**
14	Little Aston	81622	**11.7%**	**18.3%**	8.0%	0.0%	10.3%	**0.0%**	**34.2%**
15	Florence Road	73715	**34.2%**	**3.8%**	1.5%	0.0%	0.0%	**0.0%**	**52.8%**
16	Blake St	62531	**15.8%**	**24.7%**	32.2%	5.0%	1.1%	**0.0%**	**15.4%**
17	Up Reservoir Rd	63620	**24.8%**	**7.5%**	26.8%	0.5%	0.0%	**0.0%**	**35.8%**
18	Low Reservoir Rd	47371	**15.2%**	**5.2%**	40.1%	2.0%	0.0%	**0.0%**	**15.2%**
19	Percy Rd	48510	**43.1%**	**3.3%**	10.7%	0.0%	0.0%	**1.5%**	**38.9%**
20	Cradely Heath	53954	**12.4%**	**5.0%**	11.7%	10.7%	6.8%	**0.8%**	**47.5%**
21	Minworth verge	48056	**12.5%**	**16.7%**	8.9%	37.3%	5.7%	**3.3%**	**11.5%**
22	Minworth sewage	62217	**8.5%**	**11.6%**	0.0%	0.0%	0.0%	**0.0%**	**0.0%**
23	Large Bentley	141652	**15.7%**	**6.1%**	16.5%	0.0%	18.0%	**1.2%**	**0.0%**
24	Small Bentley	75088	**38.5%**	**10.5%**	0.0%	0.0%	26.0%	**8.0%**	**0.0%**
25	Hall Green Rd	154647	**12.8%**	**22.2%**	3.2%	2.8%	8.3%	**0.0%**	**25.6%**
26	Sandy Lane	82508	**24.1%**	**5.1%**	1.5%	0.0%	5.9%	**0.0%**	**23.2%**

Appendix X(a) Continued

SITE ID	SITE NAME	AREA IN BUFFER (m^2)	URB 100	SIM 100	MOWN 100	WOOD 100	SCRUB 100	PLANT 100	GARD 100
27	Vincent Srive	84865	**18.8%**	**18.2%**	0.0%	0.0%	33.3%	**0.0%**	**0.0%**
28	Crest View	46838	**33.0%**	**0.6%**	14.4%	0.0%	0.0%	**0.0%**	**49.7%**
29	Saltwells Bustop	53922	**10.1%**	**3.6%**	11.5%	7.0%	21.8%	**0.0%**	**18.3%**
30	Saltwells Verge	55073	**15.1%**	**1.7%**	19.1%	32.5%	7.8%	**0.0%**	**5.1%**
31	Soho Loop	104827	**64.3%**	**9.6%**	0.0%	0.0%	0.0%	**2.9%**	**0.0%**
32	Cole Bank Rd	51722	**16.0%**	**0.0%**	23.0%	29.9%	0.0%	**0.0%**	**22.2%**
33	Samson Quarry	81901	**0.0%**	**14.6%**	22.0%	8.6%	7.8%	**0.0%**	**0.0%**
34	Turners Hill	104953	**47.4%**	**35.5%**	3.0%	0.0%	0.0%	**0.0%**	**0.0%**
35	Burberry	66577	**30.9%**	**6.8%**	0.0%	14.2%	13.2%	**0.0%**	**0.0%**
36	Roundhay	41126	**22.2%**	**0.6%**	0.0%	0.4%	0.0%	**0.0%**	**75.3%**
37	Mid-Cole	96517	**7.3%**	**0.0%**	10.7%	0.8%	0.0%	**0.0%**	**31.1%**
38	Bearwood	42116	**19.1%**	**0.0%**	0.0%	13.2%	0.0%	**0.0%**	**66.0%**
39	Wilson Road	100915	**39.1%**	**0.4%**	4.3%	0.0%	0.0%	**0.5%**	**42.4%**
40	Ashtead Circus	68539	**68.8%**	**0.6%**	15.8%	0.6%	0.0%	**3.9%**	**1.3%**
41	Woodlands Drive	102886	**45.7%**	**16.2%**	6.9%	0.0%	0.0%	**0.0%**	**11.9%**
42	Foxyards Road	42550	**20.9%**	**0.2%**	24.1%	5.8%	2.2%	**0.0%**	**45.2%**
43	Tunnel Street	49596	**29.6%**	**0.9%**	0.9%	0.0%	11.5%	**0.0%**	**42.7%**
44	Old Park Rd	58281	**53.4%**	**2.0%**	4.3%	0.0%	0.0%	**0.0%**	**33.4%**
45	Mounts Rd	88431	**42.6%**	**8.2%**	8.5%	0.0%	8.4%	**0.6%**	**18.1%**
46	Heath Street	56153	**30.8%**	**37.7%**	7.3%	0.0%	0.0%	**2.2%**	**12.0%**
47	Walsall	60408	**46.8%**	**29.1%**	0.0%	0.0%	4.3%	**0.0%**	**6.0%**
48	Institute Rd	51755	**77.6%**	**0.1%**	0.0%	5.9%	0.0%	**0.2%**	**12.3%**
49	Tyseley Wharf	88002	**86.1%**	**0.0%**	2.5%	0.0%	0.0%	**1.6%**	**0.0%**
50	Brown Hills	59403	**6.6%**	**8.4%**	3.9%	1.0%	6.7%	**0.0%**	**30.7%**

APPENDIX XB

Urban land cover measures at 1000metre and 5000metre radii from site perimeter derived from ITE Landcover Map for the UK (Fuller et al. 1994). Suburban and Urban cover measures were added together to provide a Combined Urban cover score used in Chapters 6 & 7.

SITE ID	SITE NAME	AREA IN 1000M BUFFER (HA)	1000M SUBURB COVER	1000M URBAN COVER	AREA IN 5000M BUFFER (HA)	5000M SUBURB COVER	5000M URBAN COVER
1	Kenyan Close	325.0	49.6%	24.7%	7875.8	36.5%	10.9%
2	Frankley	344.7	46.5%	8.0%	7975.3	29.7%	3.0%
3	Erdington Large	335.4	56.8%	18.7%	7927.7	50.0%	13.5%
4	Erdington Small	325.5	55.5%	19.6%	7882.1	49.8%	13.1%
5	Glasscutters	330.8	56.4%	11.9%	7905.3	36.1%	10.9%
6	Platts Road	324.6	55.5%	14.4%	7877.8	35.5%	10.7%
7	Grange Rd	322.1	46.8%	6.6%	7862.9	48.7%	8.3%
8	Solihull Arable	350.3	23.0%	5.6%	8000.6	34.7%	5.8%
9	Ackers	325.0	32.2%	52.5%	7876.7	52.3%	24.7%
10	Sports Centre	336.9	35.4%	12.7%	7936.3	47.0%	21.5%
11	Rubery Shops	333.6	33.6%	0.5%	7920.0	24.0%	2.6%
12	Five Ways	372.7	41.2%	31.2%	8113.8	42.3%	32.9%
13	Kings Heath Pub	321.9	56.5%	3.4%	7864.8	45.3%	8.8%
14	Little Aston	362.4	24.8%	1.6%	8057.8	26.8%	2.8%
15	Florence Road	350.4	34.3%	46.0%	8003.6	44.1%	27.1%
16	Blake St	341.8	40.7%	1.8%	7958.8	24.1%	2.1%
17	Up Reservoir Rd	328.3	60.1%	25.2%	7893.7	52.2%	20.4%
18	Low Reservoir Rd	341.7	60.3%	23.2%	7963.9	52.4%	20.4%
19	Percy Rd	329.8	37.4%	51.1%	7898.3	51.7%	23.5%
20	Cradely Heath	333.9	52.4%	12.6%	7924.3	40.1%	12.6%
21	Minworth verge	328.6	34.4%	11.9%	7896.9	25.6%	5.3%
22	Minworth sewage	342.9	36.1%	7.6%	7965.8	28.1%	6.0%
23	Large Bentley Mill	397.9	51.8%	28.9%	8227.4	53.2%	18.3%
24	Small Bentley Mill	351.8	52.9%	25.3%	8005.2	52.7%	18.1%
25	Hall Green Rd	400.4	56.7%	14.3%	8236.1	46.1%	20.8%
26	Sandy Lane	357.1	55.6%	18.7%	8033.6	45.5%	17.7%
27	Vincent Srive	357.9	49.1%	21.5%	8038.6	49.1%	16.1%
28	Crest View	327.8	65.1%	2.4%	7891.0	45.0%	8.5%
29	Saltwells Bustop	333.4	43.3%	20.7%	7921.2	50.3%	17.9%
30	Saltwells Verge	335.5	36.6%	29.7%	7933.1	50.9%	17.2%

Appendix X(b) Continued...

SITE ID	SITE NAME	AREA IN 1000M BUFFER (HA)	1000M SUBURB COVER	1000M URBAN COVER	AREA IN 5000M BUFFER (HA)	5000M SUBURB COVER	5000M URBAN COVER
31	Soho Loop	377.3	36.5%	41.4%	8135.4	41.4%	32.1%
32	Cole Bank Rd	332.2	61.0%	6.0%	7911.8	53.7%	14.3%
33	Samson Quarry	354.8	45.8%	16.4%	8021.5	49.7%	24.4%
34	Turners Hill	373.1	54.1%	9.1%	8107.4	51.2%	22.5%
35	Burberry	344.1	42.4%	44.3%	7970.3	53.5%	21.2%
35	Burberry	344.1	42.4%	44.3%	7970.3	53.5%	21.2%
36	Roundhay	322.2	68.2%	7.0%	7865.9	53.3%	14.9%
37	Mid-Cole	370.3	64.2%	8.7%	8097.8	53.8%	19.0%
38	Bearwood	322.6	36.7%	27.6%	7867.8	45.2%	26.8%
39	Wilson Road	369.9	38.7%	43.8%	8098.3	46.0%	25.2%
40	Ashtead Circus	345.9	21.7%	68.7%	7979.1	42.7%	35.9%
41	Woodlands Drive	370.5	37.1%	40.6%	8094.7	47.3%	24.1%
42	Foxyards Road	324.2	56.0%	18.7%	7871.1	48.7%	19.0%
43	Tunnel Street	329.8	67.4%	10.5%	7902.8	50.0%	19.0%
44	Old Park Rd	336.9	46.9%	33.3%	7938.0	52.9%	22.6%
45	Mounts Rd	361.3	50.4%	34.3%	8056.0	48.6%	24.1%
46	Heath Street	337.6	37.0%	40.5%	7936.7	41.8%	30.3%
47	Walsall	347.3	40.0%	44.2%	7984.6	47.3%	13.5%
48	Institute Rd	331.7	47.5%	16.4%	7909.3	49.6%	15.6%
49	Tyseley Wharf	362.4	47.1%	42.8%	8061.1	54.5%	21.7%
50	Brown Hills	339.3	26.0%	6.9%	7948.9	32.1%	4.2%

APPENDIX XC

Isolation measures used in Chapters 6 & 7. Distance from derelict data was derived from Joint Development Team for West Midlands database using ARCVIEW & ARCINFO software. Distance to linear feature data was measured from Ordnance Survey Landranger series map for 1996.

SITE ID	SITE NAME	SIZE (Site area in m^2)	DISTSIM (Distance to nearest derelict site in metres)	DER100 (Derelict area in 1000m buffer in hectares)	DER5000 (Derelict area in 5000m buffer in hectares)	DISTRAIL (Distance to nearest railway in metres)	DISTRIV (Distance to nearest river in metres)
1	Kenyan Close	841.9	98	3.9	94.4	425	525
2	Frankley	4862.1	770	5.3	11.9	300	75
3	Erdington Large	5707.9	877	2.5	75.1	1400	1575
4	Erdington Small	2578.8	1042	0.0	73.5	1325	1375
5	Platt's Road	831.0	91	5.2	124.4	1125	450
6	Glasscutters	1717.7	265	6.0	139.4	900	300
7	Grange Road	348.9	2280	0.0	7.5	125	5000
8	Solihull Arable	7512.2	3591	0.0	16.1	3125	300
9	Ackers	618.3	359	29.9	50.8	25	100
10	Sports Centre	3174.3	548	15.8	35.5	0	750
11	Rubery Shops	2668.4	923	1.3	15.0	2500	200
12	Five Ways	2990.4	83	0.0	75.5	0	1375
13	Kings Heath Pub	511.5	1475	1.4	8.9	2125	75
14	Little Aston	14325.1	3729	0.0	16.7	1450	1250
15	Florence Road	5651.0	795	0.0	50.0	1200	5000
16	Blake Street	3603.1	4431	0.0	1.7	150	0
17	Low Reservoir Rd	2932.6	1248	1.5	114.8	350	2300
18	Up Reservoir Rd	1468.7	1290	1.5	112.5	575	2125
19	Percy Road	1262.4	739	17.4	46.2	975	125
20	Cradley Heath	1964.2	785	8.4	138.1	1375	450
21	Minworth Verge	1431.6	1709	0.0	4.9	50	300
22	Minworth Sewage	1449.4	821	0.0	9.6	550	250
23	Large Bentley	29401.0	0	73.8	304.6	175	0
24	Small Bentley	10489.4	88	67.5	312.6	500	0
25	Hall Green Road	38938.3	0	17.6	244.8	825	1050
26	Sandy Lane	7806.5	0	44.9	239.6	75	175
27	Vincent Drive	10124.4	0	16.2	31.3	0	125
28	Crest View	1074.3	1993	0.0	6.2	1500	300
29	Saltwells Bustop	1870.5	218	42.4	274.2	1000	5000

Appendix X(c) Continued

SITE ID	SITE NAME	SIZE (Site area in m^2)	DISTSIM (Distance to nearest derelict site in metres)	DER100 (Derelict area in 1000m buffer in hectares)	DER5000 (Derelict area in 5000m buffer in hectares)	DISTRAIL (Distance to nearest railway in metres)	DISTRIV (Distance to nearest river in metres)
30	Saltwells Verge	1343.3	106	35.7	256.8	1075	5000
31	Soho Loop	10271.3	0	2.8	72.2	0	5000
32	Cole Bank Road	1785.9	2374	0.0	21.8	300	50
33	Samsons Quarry	10548.3	493	5.9	228.9	1200	5000
34	Turners Hill	14809.4	0	2.5	240.8	1500	1875
35	Burberry	4445.3	996	14.2	43.8	775	0
36	Roundhay	614.5	0	14.6	93.0	300	900
37	Mid-Cole	8344.8	628	0.9	110.7	575	0
38	Bearwood	713.2	1133	0.0	51.7	0	5000
39	Wilson Road	11364.1	936	0.0	53.8	1725	5000
40	Ashtead Circus	4229.1	398	17.4	98.2	400	700
41	Woodlands Road	19875.8	306	4.6	136.7	75	5000
42	Foxyards	674.1	294	17.6	335.0	475	5000
43	Tunnel Street	1788.2	587	17.5	275.4	50	5000
44	Old Park Road	3446.4	0	27.2	326.4	1125	5000
45	Mounts Road	8515.8	0	12.0	312.5	0	5000
46	Heath Street	1429.1	182	2.3	72.3	0	3250
47	Walsall	3499.5	0	40.0	235.3	0	700
48	Institute Road	1968.5	532	0.2	35.4	875	2200
49	Tyseley Wharf	8662.8	99	29.3	67.9	450	850
50	Brown Hills	1920.4	222	40.3	132.7	0	425